PB87-176574

Air Quality Criteria for Particulate Matter and Sulfur Oxides (1982) (Second Addendum) Assessmet of Newly Available Health Effects Information. Final Report

(U.S.) Environmental Protection Agency
Research Triangle Park, NC

Dec 86

U.S. Department of Commerce
National Technical Information Service
NTIS

United States
Environmental Protection
Agency

Office of Health and
Environmental Assessment
Washington DC 20460

EPA/600/8-86/020F
December 1986

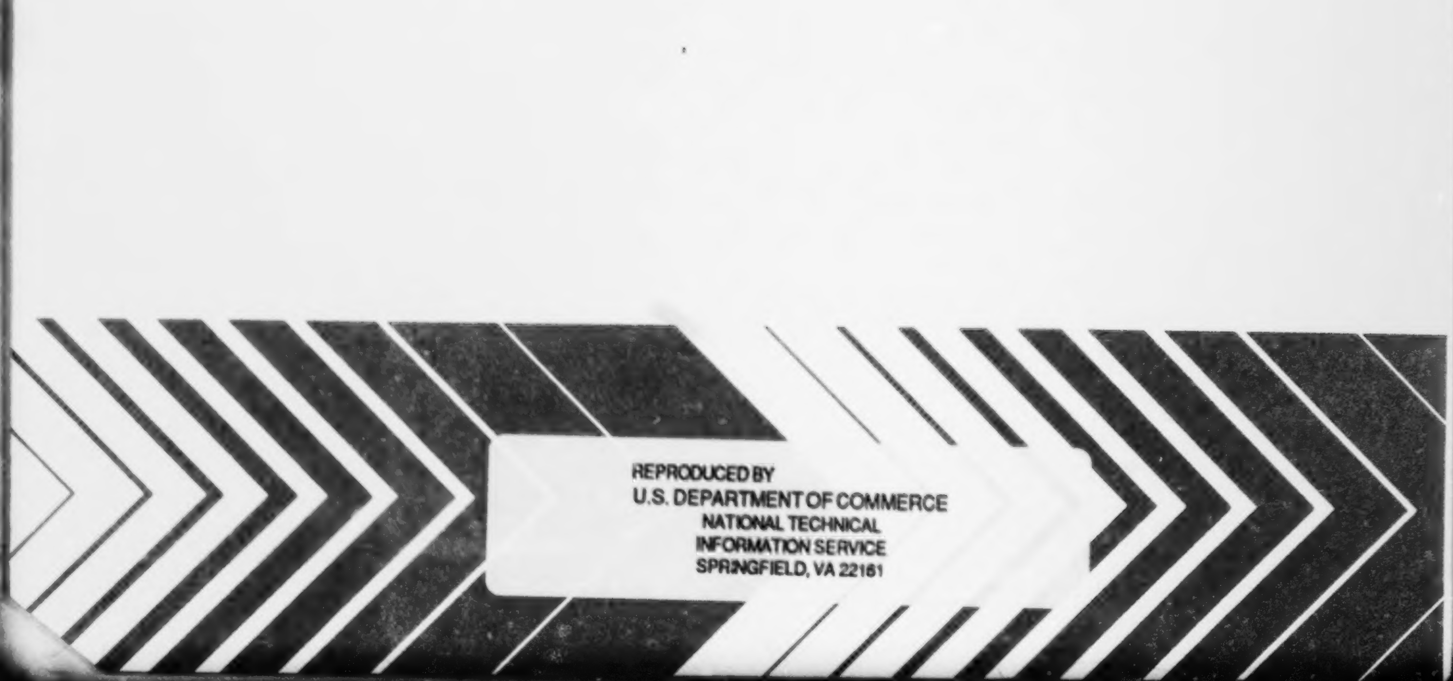
PB87-176574

Research and Development

EPA

Second Addendum to Air Quality Criteria for Particulate Matter and Sulfur Oxides (1982):

Assessment of Newly Available Health Effects Information



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TECHNICAL REPORT DATA

(Please read Instructions on the reverse before completing)

1. REPORT NO. EPA/600/8-86/020F

2.

3. RE PB87-176574

4. TITLE AND SUBTITLE
Second Addendum to *Air Quality Criteria for Particulate Matter and Sulfur Oxides (1982)*: Assessment of Newly Available Health Effects Information

5. REPORT DATE December 1986

6. PERFORMING ORGANIZATION CODE EPA/600/23

7. AUTHOR(S)
See list of authors and contributors

8. PERFORMING ORGANIZATION REPORT NO.

9. PERFORMING ORGANIZATION NAME AND ADDRESS
Environmental Criteria and Assessment Office (MD-52)
Office of Health and Environmental Assessment (ORD)
U.S. Environmental Protection Agency
Research Triangle Park, N.C. 27711

10. PROGRAM ELEMENT NO.

11. CONTRACT/GRANT NO.

12. SPONSORING AGENCY NAME AND ADDRESS
Office of Health and Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Washington, D.C. 20460

13. TYPE OF REPORT AND PERIOD COVERED Final

14. SPONSORING AGENCY CODE EPA/600/21

15. SUPPLEMENTARY NOTES

16. ABSTRACT

The second addendum to the *Air Quality Criteria for Particulate Matter and Sulfur Oxides* evaluates and assesses new scientific information that have emerged since 1982 and their implications for derivation of health-related criteria for particulate matter and sulfur oxides NAAQS. The literature through June 1986 has been reviewed thoroughly for information relevant to air quality criteria, although the document is not intended as a complete and detailed review of all literature pertaining to particulate matter and sulfur oxides.

Accordingly, the present addendum: (1) concisely summarizes key findings from the 1982 EPA criteria document and first addendum as they pertain to derivation of health-related criteria; and (2) provides an updated assessment of newly available information of potential importance for derivation of health criteria for both the particulate matter and sulfur oxides standards, with major emphasis on evaluation of human health studies published since 1982. Certain background information of crucial importance for understanding the assessed health effects findings is also concisely summarized. This includes information on physical and chemical properties of particulate matter, sulfur oxides, and associated aerosols and ambient monitoring techniques.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group

18. DISTRIBUTION STATEMENT
Release to Public

19. SECURITY CLASS *(This Report)* Unclassified

20. SECURITY CLASS *(This page)* Unclassified

21. NO. OF PAGES 217

22. PRICE

EPA Form 2220-1 (Rev. 4-77) PREVIOUS EDITION IS OBSOLETE

EPA/600/8-86/020F
December 1986

Second Addendum to Air Quality Criteria for Particulate Matter and Sulfur Oxides (1982):

Assessment of Newly Available Health Effects Information

Environmental Criteria and Assessment Office
Office of Health and Environmental Assessment
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ABSTRACT

This Addendum to the earlier 1982 U.S. EPA document, Air Quality Criteria for Particulate Matter and Sulfur Oxides, evaluates scientific information on health effects associated with exposure to various concentrations of sulfur oxides and particulate matter in ambient air. Although the literature through 1986 has been reviewed thoroughly for information relevant to air quality criteria, the present Addendum is not intended as a complete and detailed review of all literature pertaining to sulfur oxides and particulate matter. Rather, an attempt has been made to focus on the evaluation of those studies providing key information by which to delineate quantitative exposure-effect or dose-response relationships for the subject pollutants.

Although this Addendum is principally concerned with the health effects of sulfur oxides and particulate matter, other scientific data are presented and evaluated in order to provide a better understanding of these pollutants in the environment. To this end, the Addendum also includes discussions of physical and chemical properties of sulfur oxides and particulate matter; ambient air monitoring and related analytical techniques; and the respiratory tract deposition and fate associated with human exposure to the subject pollutants.

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CHAPTER 1. INTRODUCTION

The United States Clean Air Act and its 1977 Amendments mandate that the U.S. Environmental Protection Agency (U.S. EPA) periodically review criteria for National Ambient Air Quality Standards (NAAQS) and revise such standards as appropriate. The most recent periodic review of the scientific bases underlying the NAAQS for particulate matter (PM) and sulfur oxides (SO_x) culminated in the 1982 publication of the EPA document *Air Quality Criteria for Particulate Matter and Sulfur Oxides* (U.S. EPA, 1982a), an associated PM staff paper (U.S. EPA, 1982b) which examined the implications of the revised criteria for the review of the PM NAAQS, an addendum to the criteria document addressing further information on health effects (U.S. EPA, 1982c), and another staff paper relating the revised scientific criteria to the review of the SO_x NAAQS (U.S. EPA, 1982d). Based on the criteria document, addendum and staff papers, revised 24-hr and annual-average standards for PM have been proposed (Federal Register, 1984a) and public comments on the proposed revisions have been received both in written form and orally at public hearings (Federal Register, 1984b). Consideration of possible revision of the sulfur oxides NAAQS is still under way.

Since preparation of the above criteria document, addendum, and staff papers (U.S. EPA, 1982a, b, c, d), numerous new scientific studies or analyses have become available that may have bearing on the development of criteria for PM or SO_x and thus may notably impact proposed revisions of those standards now under consideration by EPA. In December 1985 the Clean Air Scientific Advisory Committee (CASAC) of EPA's Science Advisory Board met to discuss the PM proposals and possible implications of the newly available information. CASAC recommended that a second addendum to the 1982 Criteria Document (U.S. EPA, 1982a) be prepared to evaluate new studies and their implications for derivation of health-related criteria for the PM NAAQS. In the process of responding to CASAC's recommendations, the Agency also determined that it would be useful to examine studies that have emerged since 1982 on the health effects of sulfur oxides.

Accordingly, the present addendum (1) summarizes key findings from the 1982 EPA criteria document and first addendum (U.S. EPA, 1982a,c) as they pertain to derivation of health-related criteria, and (2) provides an updated assessment of newly available information of potential importance for derivation of health criteria for both the PM and SO_x standards, with major emphasis on evaluation of human health studies published since 1981. Certain background information of crucial importance for understanding the assessed health effects findings is also summarized. This includes information on physical and chemical properties of PM, sulfur oxides, and associated aerosols (including acid aerosols) and ambient monitoring techniques. However, new studies on associations between acid aerosols and health effects are being evaluated in a separate issue paper.

1.1 PHYSICAL AND CHEMICAL PROPERTIES OF AIRBORNE PARTICULATE MATTER AND AMBIENT AIR MEASUREMENT METHODS

As noted in the 1982 EPA criteria document (U.S. EPA, 1982a), airborne particles exist in many sizes and compositions that vary widely with changing source contributions and meteorological conditions. However, airborne particle mass tends to cluster in two principal size groups: coarse particles, generally larger than 2 to 3 micrometers (μm) in diameter; and fine particles, generally smaller than 2 to 3 μm in diameter. The dividing line between the coarse and the fine sizes is frequently given as 2.5 μm, but the distinction according to chemical composition is neither sharp nor fixed; it can depend on the contributing sources, on meteorology, and on the age of the aerosol.

Fine particle volume (or mass) distributions often exhibit two modes. Particles in the nuclei mode (which includes particles from 0.005 to 0.05 μm in diameter) form near sources by condensation of vapors produced by high temperature processes such as fossil-fuel combustion. Accumulation-mode particles (i.e., those 0.05-2.0 μm in diameter) form principally by coagulation or growth through vapor condensation of short-lived particles in the nuclei mode. Typically, 80 percent or more of the atmospheric sulfate mass occurs in the accumulation-mode. Particles in the accumulation mode normally do not grow into the coarse mode. Coarse particles include re-entrained surface dust, salt spray, and particles formed by mechanical processes such as crushing and grinding.

Primary particles are directly discharged from manmade or natural sources. Secondary particles form by atmospheric chemical and physical reactions, and most of the reactants involved are emitted as gaseous pollutants. In the air, particle growth and chemical transformation occur through gas-particle and particle-particle interactions. Gas-particle interactions include condensation of low-vapor-pressure molecules, such as sulfuric acid (H_2SO_4) and organic compounds, principally on fine particles. The only particle-particle interaction important in atmospheric processes is coagulation among fine particles.

As shown in Figure 1, fine atmospheric particles mainly include sulfates, carbonaceous material, ammonium, lead, and nitrate. Coarse particles consist mainly of oxides of silicon, aluminum, calcium, and iron, as well as calcium carbonate, sea salt, and material such as tire particles and vegetation-related particles (e.g., pollen, spores). The distributions of fine and coarse particles overlap; some chemical species found mainly in one mode may also be found in the other.

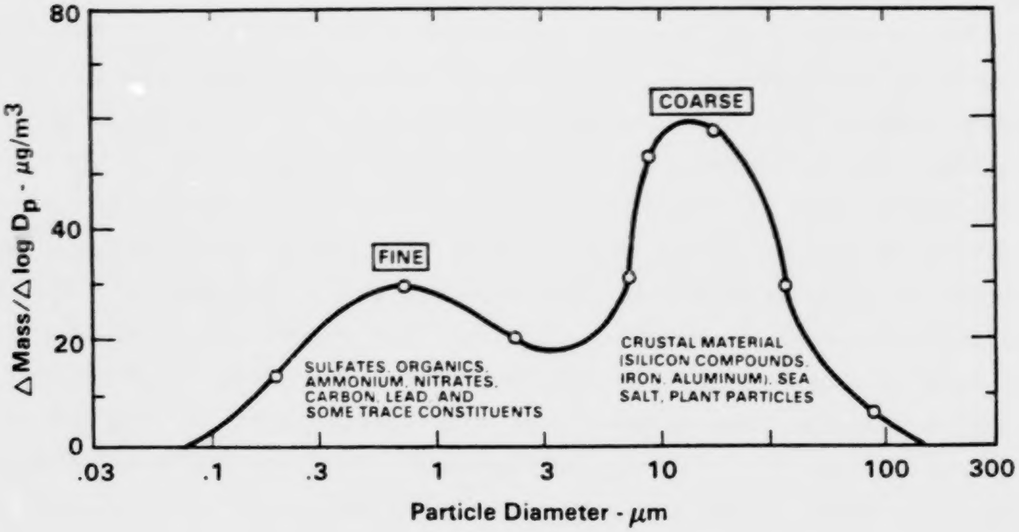

Figure 1. Representative example of typical bimodal mass distribution (measured by impactors) and chemical composition in an urban aerosol. Although some overlap exists, note substantial differences in chemical composition of fine versus coarse modes. Chemical species of each mode are listed in approximate order of relative mass contribution. Note that the ordinate is linear and not logarithmic.

Source: Modified from Whitby (1975) and National Research Council (1979).

The carbonaceous component of fine particles contains both elemental carbon (graphite and soot) and nonvolatile organic carbon (hydrocarbons in combustion exhaust and secondary organics formed by photochemistry). In many urban and nonurban areas, these species are the most abundant fine particles, after sulfates. Secondary organic particles form by oxidation of primary organics by a cycle that involves ozone and nitrogen oxides. Atmospheric reactions of nitrogen oxides yield nitric acid vapor (HNO_3) that may accumulate as nitrate particles in the fine or coarse modes. Most atmospheric sulfates and nitrates are water-soluble and tend to absorb moisture. Hygroscopic growth of sulfate-containing particles markedly affects their size, reactivity, and other physical properties which influence their biological and physical effects.

The relative proportions of particles of different chemical composition and size ranges can vary greatly in ambient air, depending upon emission sources from which they originate and interactions with meteorological conditions, e.g., relative humidity (RH) and temperature. Particles from combustion of fossil fuels or high-temperature processes, e.g., metal smelting, tend to fall in the fine (<2.5 μm) or small coarse mode (<10 μm MMD) range; those from crushing or grinding processes, e.g., mining operations, tend to be mainly in the coarse mode (>2.5 μm), with a substantial fraction in excess of 10 μm.

Another important distinction concerning airborne particles is the broad characterization that can result from different methods commonly used for routine monitoring purposes. The most commonly used methods for collection and measurement of airborne particles were described in U.S. EPA (1982a). As noted there, differences in measurements obtained from various instruments and methods used to measure PM levels have important implications for derivation of quantitative dose-response relationships from epidemiologic studies and for establishing air quality criteria and standards. It is generally not practicable to discriminate on the basis of either particle size or chemical composition when assessing particulate matter data from routine monitoring networks. Characteristics of the collected samples are dependent on the types of sources in the vicinity, weather conditions and sampling procedures. Difficulties that result and limitations of measurements were also discussed in detail in the 1982 EPA criteria document (U.S. EPA, 1982a).

When considering measurements of airborne particles it is essential to specify the method used and to recognize that results obtained with one method and under a given set of conditions are not necessarily applicable to other situations. For example, attempts have been made to relate findings based on smoke measurements (that relate mainly to dark-colored characteristics of particles from incomplete combustion of coal or other hydrocarbon fuels) to situations involving total suspended particulate matter (TSP) or size-specific fractions thereof (measured directly in terms of weight). Because the former (smoke) methods were used in many early epidemiological studies and the latter are now more often used for monitoring purposes in many countries, conversion from one type of measurement to the other would be desirable but, for reasons noted below, there can be no generally applicable conversion factor. Comparative evaluation of the two methods has been undertaken at numerous sites (Ball and Hume, 1977; Commins and Waller, 1967; Lee et al., 1972), but the results emphasize that they measure different qualities of the particulate matter and cannot be directly compared with one another (U.S. EPA, 1982a).

Sampling airborne particles is a complex task because of the wide spectrum of particle sizes and shapes. Separating particles by aerodynamic size provides a simplification by disregarding variations in particle shape and relying on particle settling velocity. The aerodynamic diameter of a particle is not a direct measurement of its size but is the equivalent diameter of a spherical particle of specific gravity which would settle at the same rate as the measured particles. Samplers can be designed to collect particles within sharply defined ranges of aerodynamic diameters or to simulate the deposition pattern of particles in the human respiratory system, which exhibits a more gradual transition from acceptance to exclusion of particles. High-volume (hi-vol) samplers, dichotomous samplers, cascade impactors, and cyclone samplers are the most common devices with specifically designed collection characteristics. These samplers rely on inertial impaction techniques for separating particles by aerodynamic size, filtration techniques for collecting the particles and gravimetric measurements for determining mass concentrations. Mass concentrations can also be estimated using methods that measure an integral property of particles such as optical reflectance, and empirical relationships between mass concentrations and the integral measurement can be used to predict mass concentration, if a valid physical model relating to the measurements exists and empirical data verify the model predictions.

The hi-vol sampler collects particles on a glass-fiber filter by drawing air through the filter at a flow rate of ~1.5 m^3/min and is used to measure total suspended particulate matter (TSP). The hi-vol sampler has cutpoints of ≈25 μm at a wind speed of 24 kph and 45 μm at 2 kph. Although sampling effectiveness is wind-speed sensitive, no more than a 10 percent day-to-day variability occurs for the same ambient concentration for typical conditions. The hi-vol is one of the most reproducible particle samplers in use, with a typical coefficient of variation of 3 to 5. One major problem associated with the glass-fiber filter used on the hi-vol is formation of artifact mass caused by the presence of acid gases in the air (e.g., artifactual formation of sulfates from SO_2), which can add 6 to 7 $\mu g/m^3$ to a 24-h sample. The hi-vol has been the sampler most widely used in the U.S. for routine monitoring and has yielded TSP mass estimates used in many American epidemiological studies.

Hi-vol samplers with size-selective inlets (SSI) have recently been developed which collect and measure particles ≤ 10 μm or ≤ 15 μm. Except for the inlet, these samplers are identical in design and operation to the TSP hi-vol. Versions are now being used in epidemiologic health effects studies, and several models are being evaluated for possible routine monitoring use.

The dichotomous sampler is a low-volume gravimetric measurement device which collects fine (≤ 2.5 μm) and coarse (≥ 2.5 μm to ≤ 10 or 15 μm) ambient particle fractions. The sampler uses Teflon® filters which minimize artifact mass formation. The earlier inlets used with this sampler were very wind-speed dependent, but newer versions are much improved. Because of low sampling flow rate, the sampler collects submilligram quantities of particles and requires microbalance analyses, but is capable of reproducibility of ±10 percent or better. The method, however, has only begun to be employed on any major scale to generate size-selective data on PM mass assessed in relation to health effects evaluated in epidemiological studies.

Cyclone inlets with cutpoints around 2 μm have long been used to separate the fine particle fraction, can be used with samplers designed to cover a range of sampling flow rates and are available in a variety of physical sizes. Applications of cyclone inlets are found in 10- and 15-μm cutpoint inlets for both dichotomous and hi-vol samplers. Samplers with cyclone inlets could be expected to have coefficients of variations similar to those of the dichotomous or SSI hi-vol samplers and, until recently, have also found only limited use in epidemiological studies of PM health effects.

Cascade impactors have been used to obtain mass distribution by particle size. Because care must be exercised to prevent errors (e.g., those due to particle bounce between stages), these samplers are normally not used as routine monitors. A study by Miller and DeKoning (1974) comparing cascade impactors with hi-vol samplers showed inconsistencies in mass collections by the impactors.

Samplers that derive mass concentrations by analytical techniques other than direct weight have been used extensively. One of the earliest was the British smokeshade (BS) sampler, which measures the reflectance of particles collected on a filter and uses empirical relationships to estimate mass concentrations. These relationships are more sensitive to carbon concentrations than mass (Bailey and Clayton, 1980) and hence are very difficult to interpret as either total or size-selective PM mass present in the atmosphere. The BS method and its standard variations typically collect PM with an ≅4.5 μm D_{50} cutpoint under field conditions, with some particles ranging from 7 to 9 μm at times being collected (McFarland et al., 1982). Thus, even if larger particles are present in the atmosphere, the BS method collects mainly fine-mode and small coarse-mode particles. The BS method neither directly measures mass nor determines chemical composition of collected PM. Rather, it measures light absorption of particles indicated by reflectance from a stain formed by particles collected on filter paper. Reflectance of light from the stain depends both on density of the stain, or amount of PM collected, and optical properties of collected PM. Smoke particles composed of elemental carbon in incomplete fossil-fuel combustion products typically make the greatest contribution to darkness of the stain, especially in urban areas. Thus, the amount of elemental carbon, but not organic carbon, in the stain tends to be most highly correlated with BS reflectance readings. Other nonblack, noncarbon particles also have optical properties which can affect the reflectance readings, but usually with negligible contribution to optical absorption.

Because the relative proportions of atmospheric carbon and noncarbon PM can vary greatly from site to site or from one time to another at the same site, the same absolute BS reflectance reading can be associated with very different amounts (or mass) of collected particles or even with very different amounts of carbon. Site-specific calibrations of reflectance readings against actual mass measurements from collocated gravimetric monitoring devices are therefore mandatory in order to obtain credible estimates of atmospheric concentrations of particulate matter based on the BS method. A single calibration curve relating mass or atmospheric concentration (in $\mu g/m^3$) of particulate matter to BS reflectance readings obtained at a given site may serve as a basis for crude estimates of the levels of PM (mainly particles <10 μm) at that site over time, so long as the chemical composition and relative proportions of elemental carbon and noncarbon PM do not change. However, the actual mass or smoke concentration at a given site may differ markedly from values calculated from a given reflectance reading on either of the two most widely used standard curves (the British and OECD standard smoke curves). Thus, much care must be taken in interpreting the meaning of any BS value reported in terms of $\mu g/m^3$, and such "nominal" expressions of airborne particle concentrations are not meaningful unless related to direct determinations of mass by gravimetric measurements carried out at the same geographical location and close in time to the BS readings.

The AISI light transmittance method is similar in approach to the BS technique, collects particles with a D_{50} cutpoint ≅5.0 μm aerodynamic diameter, uses an air intake similar to that of the BS method, and has been used for routine monitoring in some American cities. Particles are collected on a filter-paper tape periodically advanced to allow accumulation of another stain, opacity of the stain is determined by transmittance of light through the deposited material and tape, and results are expressed in terms of optical density or coefficient of haze (CoH) units per 1000 linear feet of air sampled (rather than mass units). Readings of COH units are more responsive to non-carbon particles than are BS measurements, but again, the AISI method does not directly measure mass or determine chemical composition of collected particles. Attempts to relate COH to $\mu g/m^3$ also require site-specific calibration of COH readings against mass measurements determined by a collocated gravimetric device, but the accuracy of such mass estimates are subject to question.

Since the hi-vol method collects particles much larger than those collected by BS or AISI methods, intercomparisons of PM measurements by the BS or AISI methods to equivalent TSP units, or vice versa, are very limited. For example, as shown by several studies, no consistent relationship exists between BS and TSP measurements taken at various sites or at the same site during various seasons. One exception is the relationship observed between BS and TSP during severe London air pollution episodes when low wind-speed conditions caused settling out of larger coarse-mode particles. Because fine-mode particles predominated, TSP and BS levels (in excess of ~500 $\mu g/m^3$) tended to converge, as expected if mainly fine-mode particles were present.

Another technique for determining the mass of PM collected on a filter without weighing is to quantitatively measure the attenuation by the PM sample of beta rays from a low-energy radioactive beta source. This method is very close to a true mass measurement and correlates highly with gravimetric mass concentration determinations (Lilienfeld, 1979). PM samples may be measured rapidly in the laboratory, or instruments incorporating both the sampling apparatus and the beta source and detector are available for automatic, on-site measurements. Sample periods typically range from 0.5 to 2 hours, and quasi-continuous PM monitoring can be achieved in the instrument by configuring the beta detector to monitor the particle collection area of the filter as the particles accumulate. Various particle size-selective inlets can be used with a beta attenuation instrument to effect size-limited PM measurements. The technique has generally good precision, but it is subject to errors from detector drift and absorption of moisture from the atmosphere by the filter material or the collected PM (Lilienfeld, 1985).

The piezoelectric microbalance is a device that measures PM continuously by electronically measuring the change in the resonant frequency of a quartz crystal as PM is deposited on its surface, either by impaction or electrostatic precipitation. Although very sensitive, this technique is subject to measurement error from imperfect adhesion of particles to the crystal, moisture and temperature dependence, and sensitivity to certain pollutant gases (Lilienfeld, 1985). In addition, the dynamic range of the technique is limited, and the crystal must be cleaned frequently.

Another resonant frequency technique is the tapered element oscillating microbalance (TEOM), which continuously measures the mass of the PM collected on a filter mounted on the end of a cantilevered element by electronically measuring the change in the element's resonant frequency of oscillation. Because the PM is collected on a filter, this method is more rugged than the piezoelectric microbalance and compares more favorably with other filtration methods (Lilienfeld, 1985). However, it is less sensitive, suffers from the same potential interference from absorption of moisture by the filter and collected PM, and requires that the filter be changed periodically.

The integrating nephelometer, an optical instrument which measures light scattered by suspended particles, can be used to continuously measure ambient concentrations of PM without collection, but such measurements are only indirectly related to the mass concentration of the particles. Light scattering varies with particles size and is maximum in the 0.3 to 0.8 µm (accumulation mode) size range (Charlson et al., 1978). Thus, nephelometer measurements are most useful for fine particles and visibility monitoring and correlate poorly with broad-size-range PM measurements unless the ambient particle size distribution is dominated by fine particles. Heating the sampled air to reduce its relative humidity is necessary to minimize the effect of high humidity on particle size.

Many analytical techniques are available to determine chemical properties of particles collected on a suitable substrate. Most of the techniques, such as those for elemental sulfur, have been shown to be more precise than the analyses for gravimetric mass concentration. Methods are available that provide reliable analyses for sulfates, nitrates, organic fractions, and elemental composition (e.g., sulfur, lead, silicon), but not all analyses can be used for all particle samples because of factors such as incompatible substrates or inadequate sample size. Results can be misinterpreted when samples have not been appropriately segregated by particle size and when artifact mass is formed on the substrate rather than collected in particulate form, e.g., positive artifacts likely in nitrate and sulfate determinations (as noted below).

1.2 PHYSICAL/CHEMICAL PROPERTIES OF SULFUR OXIDES AND THEIR TRANSFORMATION PRODUCTS AND AMBIENT MEASUREMENT METHODS

The only sulfur oxide that occurs at significant concentrations in the atmosphere is sulfur dioxide, one of the four known gas-phase sulfur oxides (sulfur monoxide, sulfur dioxide, sulfur trioxide, and disulfur monoxide). As discussed in U.S. EPA (1982a), sulfur dioxide is a colorless gas detectable by taste at levels of 1000 to 3000 $\mu g/m^3$ (0.35-1.05 ppm). Above 10,000 $\mu g/m^3$ (3.5 ppm), it has a pungent irritating odor.

As also discussed in U.S. EPA (1982a), SO_2 is mainly removed from the atmosphere by gaseous, aqueous, and surface oxidation to form acidic sulfates. Gas-phase oxidation of SO_2 by the hydroxyl (OH) radical is well understood; not so well understood, however, is oxidation of SO_2 by hydroperoxyl (HO_2) and methyl peroxyl (CH_3O_2) radicals. The ready solubility of SO_2 in water is due mainly to formation of bisulfite (HSO_3^-) and sulfite (SO_3^{2-}) ions, which are easily oxidized to form acidic sulfates by reacting with catalytic metal ions and dissolved oxidants. Sulfur dioxide reacts on the surface of a variety of airborne solid particles, such as ferric oxide, lead dioxide, aluminum oxide, salt, and charcoal.

Sulfur trioxide (SO_3), which can be emitted into the air directly or result from reactions mentioned earlier, is a highly reactive gas. In the presence of moisture in the air, it is rapidly hydrated to form sulfuric acid. In the air, then, it is sulfuric acid in the form of an aerosol that is found rather than SO_3, and it is generally associated with other pollutants in droplets or solid particles of widely varying sizes. The acid is strongly hygroscopic, and droplets containing it readily take up further moisture from the air until they are in equilibrium with their surroundings. If any ammonia is present, it reacts with sulfuric acid to form various ammonium sulfates, which continue to exist as an aerosol (in droplet or crystalline form, depending on the relative humidity).

The sulfuric acid may also react further with other compounds in the air to produce other sulfates. Some sulfates reach the air directly from combustion or industrial sources, and near oceans, sulfates exist in aerosols generated from ocean spray. As discussed in U.S. EPA (1982a), sulfate particles fall mainly in the fine-mode (<2.5 μm) size range. These particles, in the presence of moisture in air, combine with water to form coarse-mode aerosols (i.e., >2.5 μm).

Many sulfur compounds are present in the complex mixture of urban air pollutants. Some are naturally occurring and some are manmade. Total biogenic sulfur emissions in the United States have been estimated to be in the range of 5 to 6 million metric tons annually. Additional contributions from coastal and oceanic sources may also be significant. Anthropogenic (manmade) sources are estimated to emit about 26 to 27 million metric tons of SO_x (mostly SO_2) annually in the United States. Most manmade sulfur oxide emissions are from stationary point sources; over 90 percent of these are SO_2 and the rest are sulfates.

Once SO_2 is emitted into the lower atmosphere, maintenance of a tolerable environment depends on the ability of wind and turbulence to disperse the pollutants. Factors affecting the dispersion of SO_2 from combustion sources include (1) temperature and efflux velocity of the gases, (2) stack height, (3) topography and the proximity of other buildings, and (4) meteorology. Some of the SO_2 emitted into the air is removed unchanged onto various surfaces, including soil, water, grass and vegetation. The remaining SO_2 is transformed into sulfuric acid or other sulfates by various processes in the presence of moisture, and these transformation products are then removed by dry deposition processes or by precipitation. The relative proportion of SO_2 and its transformation products resulting from atmospheric processes varies with increasing distance from emission sources and residence time (age) in the atmosphere. With long-range transport (over hundreds or thousands of kilometers), extensive transformation of SO_2 to sulfates occurs, with dry deposition of acidic sulfates or their wet depositon in rain or snow contributing to acidic precipitation processes.

The most commonly used collection and measurement methods for sulfur oxides were described in the 1982 EPA criteria document (U.S. EPA, 1982a). A clear understanding of the underlying bases and limitations of particular methods is essential for adequate interpretation of epidemiological studies discussed later. If SO_2 were the only contaminant in air, all measurement methods for that gas would give comparable results, indicating the true concentration of SO_2. In typical urban environments, however, other pollutants are always present and although sampling procedures can be arranged to minimize interference from particulate matter by first filtering the air, errors still arise due to other gases and vapors. Thus, variations in specificity and accuracy of methods must be taken into account in comparing results from various studies.

Methods for measurement of SO_2 include (1) manual methods, which involve collection of the sample over a specified time period and subsequent analysis by a variety of analytical techniques, and (2) automated methods, in which sample collection and analysis are performed continuously and automatically. In the most commonly used manual methods, the analyses of the collected samples are based on colorimetric, titrimetric, turbidimetric, gravimetric, x-ray fluorescent, chemiluminescent, and ion exchange chromatographic measurement principles.

The most widely used manual method for determination of atmospheric SO_2 is the West-Gaeke pararosaniline method. An improved version of this colorimetric method, adopted in 1971 as the U.S. EPA reference method, can measure ambient SO_2 at levels as low as 25 $\mu g/m^3$ (0.01 ppm) with 30 min to 24 hr sampling time. The method has acceptable specificity for SO_2, if properly implemented; however, samples collected in tetrachloromercurate(II) can undergo temperature-dependent decay leading to the underestimation of ambient SO_2 concentrations.

A variation of the method uses a buffered formaldehyde solution for sample collection, reducing the temperature-dependent decay problem. Certain American epidemiological studies employed the West-Gaeke or other variations of the pararosaniline method.

A titrimetric (acidometric) method, whereby SO_2 is collected in dilute hydrogen peroxide and the resultant H_2SO_4 is titrated with standard alkali, is the standard method mainly used in Great Britain and by the Organization for Economic Cooperation and Development (OECD). The method requires long sampling times (24 h), is subject to interference from atmospheric acids and bases, and can be affected by errors due to evaporation of reagent during sampling, titration errors, and alkaline contamination of glassware. It has been used to provide aerometric SO_2 estimates reported in many British and European epidemiological studies.

Some other methods use alkali-impregnated filter papers for collection of SO_2 and subsequent analysis as sulfite or sulfate. Most involve extraction prior to analysis; but nondispersive x-ray fluorescence allows direct measurement of SO_2 collected on sodium carbonate-impregnated membrane filters. These methods have not been widely used for routine air monitoring or epidemiological studies.

Two of the most sensitive methods for measuring SO_2 are based on chemiluminescence and ion exchange chromatography. With the former, SO_2 is absorbed in a tetrachloromercurate solution and then oxidized with potassium permanganate; oxidation of the absorbed SO_2 is accompanied by chemiluminescence detected by a photomultiplier tube. With the latter, ion exchange chromatography can be used to determine ambient levels of SO_2 absorbed into dilute hydrogen peroxide and oxidized to sulfate, or SO_2 absorbed into a buffered formaldehyde reagent. These methods have not yet been widely employed for routine monitoring uses.

Sulfation methods, based on reaction of airborne sulfur compounds with lead dioxide paste to form lead sulfate, have been used both in the United States and Europe to estimate ambient SO_2 concentrations over extended time periods. However, data obtained by sulfation methods are affected by many physical and chemical variables and other interferences (such as wind speed, temperature, and humidity); and they are not specific for SO_2, since sulfation rates are also affected by other airborne sulfur compounds (e.g., as sulfates). Thus, although sulfation rates (mg SO_3/100 cm^2/day) have been converted to rough estimates of SO_2 levels (in ppm), these cannot be accepted as accurate measurements of atmospheric SO_2 levels. This is notable here because lead dioxide gauges provided estimates of SO_2 data used in some pre-1960s British epidemiological studies and also in some American epidemiologic studies.

Automated methods for measuring ambient SO_2 levels have been widely used for air monitoring. Some early continuous SO_2 analyzers, based on conductivity and coulometry, were subject to interference by many ambient air substances. More recent commercially available analyzers using these measurement principles exhibit improved specificity for SO_2 through incorporation of sophisticated chemical and physical scrubbers.

Continuous SO_2 analyzers that use flame photometric detection (FPD), fluorescence, or second-derivative spectrometry are now commercially available. The FPD method involves measurement of the band emission of excited SO_2 molecules formed from sulfur species in a hydrogen-rich flame and can exhibit high sensitivity and fast response, but must be used with selective scrubbers or coupled with gas chromatographs to achieve high specificity. Fluorescence analyzers detect characteristic fluorescence of the SO_2 molecule when irradiated by UV light, have acceptable sensitivity and response times, are insensitive to sample flow rate, and require no support gases. However, they can be affected by interference due to water vapor (quenching effects) and certain aromatic hydrocarbons and must employ ways to minimize such effects. Second-derivative spectrometry can provide highly specific measurement of SO_2 in the air, with continuous analyzers based on this principle being insensitive to sample flow rate and requiring no support gases. U.S. EPA has designated continuous analyzers based on many of the above principles (conductivity, coulometry, flame photometry, fluorescence, and second-derivative spectrometry) as equivalent methods for measurement of atmospheric SO_2.

Two main methods have been used to measure total water-soluble sulfates collected on filters along with other suspended particulate matter. With the turbidimetric method, samples are collected on sulfate-free glass fiber or other efficient filters, the sulfate is extracted and precipitated with barium chloride, and the turbidity of the suspension is measured spectrophotometrically. Samples are normally collected over 24-h periods by hi-vol sampler. However, no distinction can be made between sulfates and sulfuric acid present in the air and collected on the filters; and some material present as acid in the air may be converted to neutral sulfate on the filter during sampling. With the methylthymol blue method, samples are collected as in the turbidimetric method and the extract is reacted with barium chloride, but the barium remaining in solution is then reacted with methylthymol blue and the sulfate determined colorimetrically by measurement of uncomplexed methylthymol blue. This modification allows the procedure to be automated, but the same limitations as noted for the turbidimetric method apply, including lack of distinction between sulfates and sulfuric acid.

As for sulfuric acid, no fully satisfactory method exists for its measurement in the presence of other pollutants in the air, but some procedures exist for examining acidic properties of suspended particles or acid aerosols in general. Almost all of the strong acid content of ambient aerosols consists of sulfuric acid (H_2SO_4) and its partial atmospheric neutralization product, ammonium bisulfate (NH_4HSO_4); however, ammonium sulfate [$(NH_4)_2SO_4$], the final neutralization product, is only weakly acidic. Nitric acid (HNO_3) and hydrochloric acid (HCl) are other strong acids found in the ambient air (mainly as vapors or, when incorporated into fog droplets, as constituents of acid aerosols). Ambient air acidic aerosol concentrations can be expressed in terms of µmols H^+/m^3 or as H_2SO_4 equivalent in $\mu g/m^3$ (at 98 µg/µmol). Unfortunately, no systematic surveys of average acid aerosol concentrations in United States airsheds were available at the time the 1982 EPA criteria document (U.S. EPA, 1982a) was prepared, nor is such systematic survey information available for more current acidic aerosol levels. However, Lioy and Lippmann (1986) have recently summarized some of the highest levels reported for recent years in North America, including levels in the range of 20 to 30 $\mu g/m^3$ H_2SO_4 (1 hr mean). This is in contrast to the highest level (680 $\mu g/m^3$ H_2SO_4 1 hr mean) recorded in the United Kingdom in London in 1962 and even higher levels almost certainly present during earlier London air pollution episodes.

1.3 KEY AREAS ADDRESSED IN EMERGING NEW HEALTH EFFECTS DATA

Important new health effects information has emerged in three main areas since preparation of the 1982 EPA criteria document and addendum: (1) new data which permit more definitive characterization of respiratory tract deposition patterns for inhaled particles of various size ranges, e.g., fine-mode (<2.5 µm) vs. larger coarse mode particles (>2.5 µm, <10 µm, <15 µm, etc.); (2) new reanalyses of certain key British epidemiology studies, which used BS methods for measuring PM levels, and additional new epidemiologic studies, employing other non-gravimetric or gravimetric PM measurement methods, that assess health effects associated with exposures to PM and SO_x in contemporary urban airsheds of the 1970s and 1980s; and (3) new controlled human exposure studies which more precisely define exposure-response relationships for pulmonary function decrements and respiratory symptoms due to acute SO_2 exposure.

CHAPTER 2. RESPIRATORY TRACT DEPOSITION AND FATE

2.1 RESPIRATORY TRACT DEPOSITION AND FATE OF INHALED AEROSOLS

As discussed in U.S. EPA (1982a), the respiratory system is the major route of human exposure to airborne suspensions of particles (aerosols) and gases such as SO_2. In inhalation toxicology, deposition refers to removal from inspired air of inhaled particles or gases by the respiratory tract and the initial regional pattern of these deposited materials. Clearance refers to subsequent translocation (movement of material within the lung or to other organs), transformation, and removal of deposited substances from the respiratory tract. It can also refer to removal of reaction products formed from SO_2 or particles. Retention refers to the temporal pattern of uncleared deposited particulate materials or gases and reaction products. These phenomena are complicated by interactions that occur among particles, gases such as SO_2 or endogenous ammonia, and water vapor in the airways.

Deposition patterns of inhaled aerosols and gases are affected by physical and chemical properties, e.g., aerosol particulate size distribution, density, shape, surface area, electrostatic charge, hygroscopicity or deliquescence, chemical composition, gas diffusivity and solubility, and related reactions. The geometry of the respiratory airways from nose and mouth to the lung parenchyma also influences aerosol deposition; important morphological parameters include diameters, lengths, inclinations to vertical, and branching angles of airway segments. Physiological factors that affect deposition include breathing patterns, respiratory tract airflow dynamics, and variations of relative humidity and temperature in the airways. Clearance from the respiratory tract depends on many factors, including site of deposition, chemical composition and properties of deposited particles, reaction products, mucociliary transport in the tracheobronchial tree, macrophage phagocytosis, and pulmonary lymph and blood flow. An understanding of respiratory tract anatomy and regional deposition and clearance of particles is essential for interpretation of the results of health effects studies discussed later.

The respiratory tract includes the passages of the nose, mouth, nasopharynx, oropharynx, epiglottis, larynx, trachea, bronchi, bronchioles, and small ducts and alveoli of the pulmonary acini. In regard to respiratory tract deposition and clearance of inhaled aerosols, three main regions can be considered: (1) the extrathoracic (ET) region, which includes the airways extending from the nares down to the epiglottis and larynx at the entrance to the trachea (the mouth is included in this region during mouth breathing); (2) the tracheobronchial (TB) region, which includes the primary conducting airways of the lung from the trachea to the terminal bronchioles (i.e., that portion of the lower respiratory tract having a ciliated epithelium); and (3) the pulmonary (P) region, which consists of the parenchymal airspaces of the lung, including the respiratory bronchioles, alveolar ducts, alveolar sacs, atria, and alveoli (i.e., the gas-exchange region). The extrathoracic region, as defined above, corresponds exactly to the nasopharynx, as defined by the International Commission on Radiological Protection (ICRP) (Task Group on Lung Dynamics, 1966). The thoracic region corresponds to that portion of the respiratory tract distal to, and including, the trachea (i.e., TB + P).

As discussed in U.S. EPA (1982a), evaluation of mechanisms by which inhaled particles ultimately affect human health requires recognition of the importance of deposition and clearance phenomena in the respiratory tract. Major regions of the respiratory tract differ markedly in structure, size, function, and sensitivity or reactivity to deposited particles. They also have different mechanisms for particle elimination or clearance.

The 1982 EPA criteria document depicted available experimental deposition data for total and regional deposition in a series of figures (i.e., Figures 11-3 to 11-9 of U.S. EPA, 1982a). Curves for alveolar deposition and estimates of tracheobronchial deposition, along with an extrapolation of the upper bound of the TB curve to the point predicted by Miller et al. (1979), are reproduced here in Figure 2. Added to the figure are the more recent data of Svartengren (1986), Heyder (1986), and Emmett et al. (1982) for deposition of particles ≥ 10 µm in aerodynamic diameter (D_{ae}) in healthy adult subjects breathing through a mouthpiece.

In the studies reported by Heyder (1986), mean inspiratory flow rates of 250 and 750 $cm^3 s^{-1}$ were used with a four-second breathing cycle, resulting in minute ventilations of 7.5 and 22.5 L min^{-1}, respectively. At the higher flow rate, TB deposition of 10 µm D_{ae} particles was 0.14; fractional deposition for

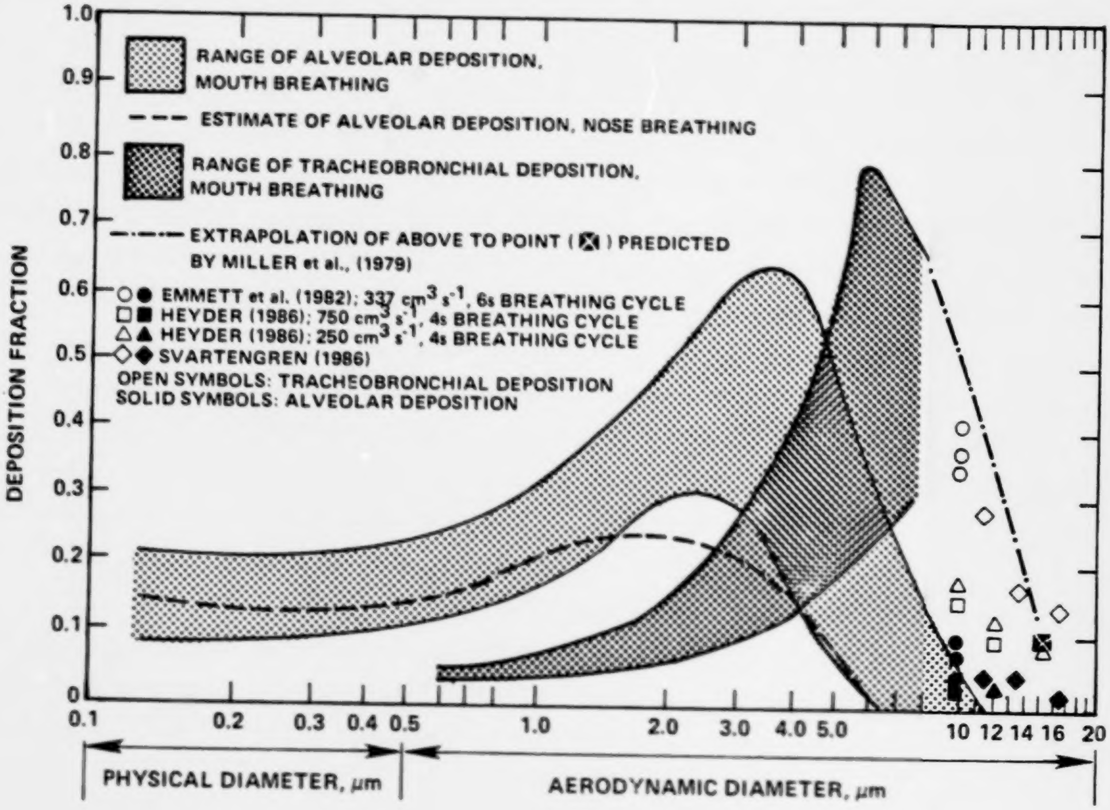

Figure 2. Regional deposition of monodisperse aerosols by indicated particle diameter for mouth breathing (alveolar, tracheobronchial) and nose breathing (alveolar). The alveolar band indicates the range of results found by different investigators using different subjects and flow parameters for alveolar deposition following mouth breathing. Variability is also expected following nasal inhalation. The tracheobronchial band indicates intersubject variability in deposition over the range of sizes as measured by Chan and Lippmann (1980). Deposition is expressed as fraction of particles entering the mouth (or nose). Also shown is an extrapolation of the upper bound of the TB curve to the point predicted by Miller et al. (1979). The extrapolation illustrates the likely shape of the curve in this size range but is uncertain. However, the data of Emmett et al. (1982), Heyder (1986), and Svartengren (1986) tend to substantiate this extrapolation. In the Svartengren (1986) studies, subjects took maximally deep inhalations at a flow of 500 cm^3 s^{-1}.

12 µm D_{ae} particles was 0.09. In contrast, the lower flow rate yielded deposition fractions of 0.17 and 0.12, respectively, for 10 µm and 12 µm D_{ae} particles. Emmett et al. (1982) observed an average TB deposition of 0.36 in three subjects who inhaled 10 µm D_{ae} particles at a mean inspiratory flow rate of 337 cm^3s^{-1} with 10 breaths/min (i.e., minute ventilation of 10.1 L min^{-1}). Under these breathing conditions the alveolar region deposition fraction for 10 µm particles averaged 0.06.

The deposition of 11.5, 13.7, and 16.4 µm D_{ae} particles was studied by Svartengren (1986) using a different exposure regime. Subjects took four maximally deep inhalations at a flow of 500 cm^3s^{-1} from a glass bulb apparatus each time particles were sprayed up into the bulb. Exposure times varied from 2 to 5 min. Six subjects were studied at the 11.5 and 13.7 µm sizes, while five subjects were studied at 16.4 µm D_{ae}. The average alveolar deposition fraction was 0.01 at the largest particulate size and 0.04 at the 11.5 and 13.7 µm sizes. By subtracting alveolar deposition from the measured total lung deposition, the average TB deposition fractions of the 11.5, 13.7, and 16.4 µm D_{ae} particles were 0.27, 0.17, and 0.12, respectively. The data of Svartengren (1986), along with the data of Heyder (1986) and Emmett et al. (1982), tend to substantiate the extrapolation of the upper bound of the TB curve in Figure 2 to the point predicted by Miller et al. (1979).

Numerous subject-related and environmental factors can influence deposition and clearance of aerosols, including inhalation patterns (rate and route), airway dimensions in relation to pulmonary function measurements, disease state, particle composition, and the presence of pollutant gases. Detailed discussion of effects of such factors on deposition patterns is beyond the scope of this addendum (for more details, see U.S. EPA, 1982a,b; Lippmann et al., 1980; Garrard et al., 1981; Svartengren et al., 1986; Lippmann and Schlesinger, 1984). However, the results of Heyder et al. (1982) on the biological variability of particulate deposition in controlled and spontaneous mouth breathing are of interest since this was an important issue raised in the 1982 EPA criteria document. Using both breathing patterns and particulate sizes ranging from 1 to 7µm D_{ae}, they studied total deposition and deposition rate in 20 subjects. The variability of deposition rate between subjects spontaneously breathing the same aerosol is associated with morphological and physiological factors but is mainly governed by physiological factors (i.e., primarily individual flow rate). Heyder et al. (1982) contend that this type of variability is the most important when considering health-related issues of inhaled particulate matter.

Data on respiratory tract deposition can be used to provide an evaluation of deposition of typically observed ambient particulate distributions. The similarity of experimental deposition data from human subjects breathing monodisperse aerosols in a laboratory setting to the general population breathing multimodal urban aerosols was examined in studies published after preparation of the 1982 EPA criteria document (U.S. EPA, 1982a). Hiller et al. (1982) studied total respiratory tract deposition in five subjects using a mixture of monodisperse polystyrene latex spheres 0.6, 1, and 2 μm in size. Their experimental results suggest that the deposition of mixed monodisperse and monodisperse single aerosols is similar for fine particles. However, the theoretical modeling of Diu and Yu (1983) indicate that the regional deposition patterns of polydisperse aerosols can be quite complex. They assumed a log normal size distribution and studied total and regional deposition with nasal and mouth breathing for geometric standard deviations (σ_g) of 1.0 (monodisperse), 1.5, 2.5, and 3.5. The results of Diu and Yu (1983) are consistent with the observation of Morrow (1984) that the mass deposition of mono- and polydisperse aerosols differs little if $\sigma_g < 2$. Typically, σ_g values reported for distribution of urban and rural aerosols is usually around 2 (see Chapter 5, U.S. EPA, 1982a). In the theoretical studies of Diu and Yu (1983), larger values of σ_g are predicted to impart significant complexities in regional deposition patterns due to competing mechanisms interacting with the sequential filtering effect of the respiratory tract.

Over half of the total mass of a typical ambient mass distribution would be deposited in the extrathoracic region, most of this being coarse particles, during normal nasal breathing (see Chapter 11 of U.S. EPA, 1982a). Clearance of most of this material to the esophagus would occur within minutes. Some fraction of the hygroscopic fine mass (e.g., sulfates and nitrates that grow to 2-4 μm in the respiratory tract) might also be deposited and dissolve in the extrathoracic region. Smaller fractions of both the hygroscopic and nonhygroscopic fine particles (mostly <1 μm) would be deposited in the tracheobronchial and alveolar regions, respectively. Clearance of hygroscopic material by dissolution and reaction would be relatively rapid in both regions. Clearance of insoluble coarse-mode substances would increase from less than an hour for the larger particles deposited in the upper portion of the tracheobronchial region to as much as a day for that deposited more distally. Insoluble fine and coarse particles deposited in the alveolar region have clearance half-times varying from weeks to years for the fast phase and slow phase, respectively.

With mouth-only breathing, the regional deposition pattern changes markedly, with extrathoracic deposition reduced and both tracheobronchial and pulmonary deposition enhanced. Extrathoracic deposition, although reduced, still would be dominated by coarse mode aerosols and contain little fine-mode contribution. Endogenous ammonia in human airways may, however, reduce the deposition of acid aerosols (U.S. EPA, 1982b). Remaining non-hygroscopic fine particle deposition efficiency would change little over nasal breathing (<20 percent).

In essence, regional deposition of ambient particles in the respiratory tract does not occur at divisions clearly corresponding to atmospheric aerosol distributions. Coarse-mode and hygroscopic fine-mode particles are deposited in all three regions. A fraction (5 to 25 percent) of the remaining fine-mode particles (e.g., organics and carbon not associated with hygroscopic material) is deposited in the tracheobronchial/alveolar regions. With mouth-only breathing, as illustrated in Figure 2, little particulate mass in excess of 15 μm is deposited in the thoracic region, and little mass greater than 10 μm is deposited in the alveolar region.

Oronasal breathing (partly via the mouth and partly nasally) typically occurs for healthy adults while undergoing moderate to heavy exercise. Swift and Proctor (1982) computed deposition for oronasal breathing as a function of particulate size, correcting for deposition in the parallel nasal and oral airways, and compared these results to those for mouth breathing via tube. Using minute ventilations of 24.5 and 15 Lmin^{-1}, their analyses predicted that total thoracic deposition at all sizes is more or less essentially the same as for pulmonary deposition noted above for mouth only breathing, i.e., with very few particles over 10 μm D_{ae} in size being likely to reach tracheobronchial regions. Tracheobronchial deposition with oronasal breathing at a higher minute ventilation (45 Lmin^{-1}) has been examined by Miller et al. (1984). Data for extrathoracic and tracheobronchial deposition were fit to logistic regression models yielding significantly improved fits of the deposition data. As done by Swift and Proctor (1982), a 50/50 split in airflow between the nasal and oral pathways was assumed. Simulated oronasal breathing at a minute ventilation of 45 Lmin^{-1} resulted in tracheobronchial deposition fractions of 0.21, 0.17, 0.14 and 0.09 for particles of 8, 9, 10, and 12 μm in aerodynamic diameter, respectively. When the experimental deposition data of Heyder (1986), separately for nasal and oral breathing, are combined to simulate oronasal breathing, the results are in agreement with the analyses of Miller et al. (1984). Bowes and Swift (1986) studied mouth deposition during natural oronasal breathing and found that 58% of 10 μm particles and 78% of 15 μm particles deposited in the mouth. Nasal deposition efficiencies were, however, not measured.

More recently, thoracic deposition and its component parts have been examined by Miller et al. (1986), as a function of particulate size, for ventilation rates ranging from normal respiration to heavy exercise in individuals who, as per Niinimaa et al. (1981), habitually breathe oronasally (mouth breathers) and in those who normally employ oronasal breathing when minute ventilations exceed about 35 Lmin^{-1} (normal augmenters). Published data from various laboratories for ET and TB deposition, along with previously unpublished data of Lippmann and co-workers at New York University, were fit to logistic regression models prior to examining the influences of breathing mode and activity level on TB, P, and thoracic (TB + P) deposition. For the ET region, an impaction parameter was used that was a function of aerodynamic diameter and inspiratory flow rate, and the logistic models provided significantly improved fits of the nasal and oral inspiration data compared to the linear models of Yu et al. (1981) that also used an impaction parameter and that formed the basis of the Swift and Proctor (1982) analyses. Since TB deposition is due primarily to inertial impaction in the upper airways and to sedimentation in the lower airways, the logistic analysis for the TB region was based upon aerodynamic diameter rather than on an impaction parameter. The proportionality of airflow between the nose and mouth as a function of activity level was determined from Figure 2 of Niinimaa et al. (1981).

Thoracic deposition results given by Miller et al. (1986) are shown in Figure 3, along with the thoracic deposition results of Swift and Proctor (1982). With minute ventilations ($\dot{V}_E$) of 40 or 60 Lmin^{-1} (panel A), there is not much difference between normal augmenters and mouth breathers in thoracic deposition for D_{ae} beyond the peak of the deposition curve. For $\dot{V}_E$ less than 35 Lmin^{-1}, the Miller et al. (1986) analyses result in substantially lower deposition in normal augmenters compared to mouth breathers. As $\dot{V}_E$ increases, thoracic deposition for normal augmenters initially decreases for a given D_{ae}, increases through the oronasal switching point, and then decreases. For mouth breathers, however, there are minimal changes in thoracic deposition at lower ventilation rates with monotonic declines in deposition as $\dot{V}_E$ increases beyond 30 $Lmin^{-1}$.

Swift and Proctor (1982) computed bands of total thoracic deposition as a fraction of particles entering the mouth and nose during oronasal and oral breathing, using $\dot{V}_E$ of approximately 24.5 $Lmin^{-1}$ and 15 $Lmin^{-1}$, respectively. The shaded area of Panel B (Figure 3) represents a composite of these data based on the lower band of the low $\dot{V}_E$ and the upper band of the higher $\dot{V}_E$. While neither Swift and Proctor (1982) nor the U.S. EPA (1982a,b) extended the bands for TB deposition beyond 8 μm, some thoracic deposition could be projected for 10 to 15 μg particles with oronasal breathing. More recent experimental data utilized in Miller et al. (1986) indicate that there is a gradual decline in thoracic deposition for large particulate sizes and that there can be significant deposition of particles greater than 10 μm, particularly for mouth breathers.

It should be noted that the deposition studies cited previously all used adult subjects, yet many of the epidemiology studies cited in the PM/SO_x criteria document (U.S. EPA, 1982a) and in this addendum report effects observed in children. Anatomical and functional differences between adults and children are likely to yield complex interactions with the major mechanisms affecting respiratory tract deposition. In a study of over 1800 Mexican-American, white, and black children 7 to 20 years of age, Hsu et al. (1979) found significant differences of lung volume and flow rate among the three races, and between male and female subjects. Further analyses of these data by Hsi et al. (1983) demonstrated that using sitting height as a predictor greatly reduced the racial differences of ventilatory function and allowed the application of a single set of prediction equations for children of all three groups. Other studies are available on normal pulmonary function values (de Swiniarski et al., 1982), intrasubject variability (Hutchinson et al., 1981), influence of physical performance capacity on the growth of lung volumes (Anderson et al., 1984), and postnatal growth and size of the pulmonary acinus (Osborne et al., 1983).

To date, experimental deposition data in children's lungs are not available. Analogous to the development of mathematical models for deposition in adults, the thrust for age-dependent dosimetry modeling has been from

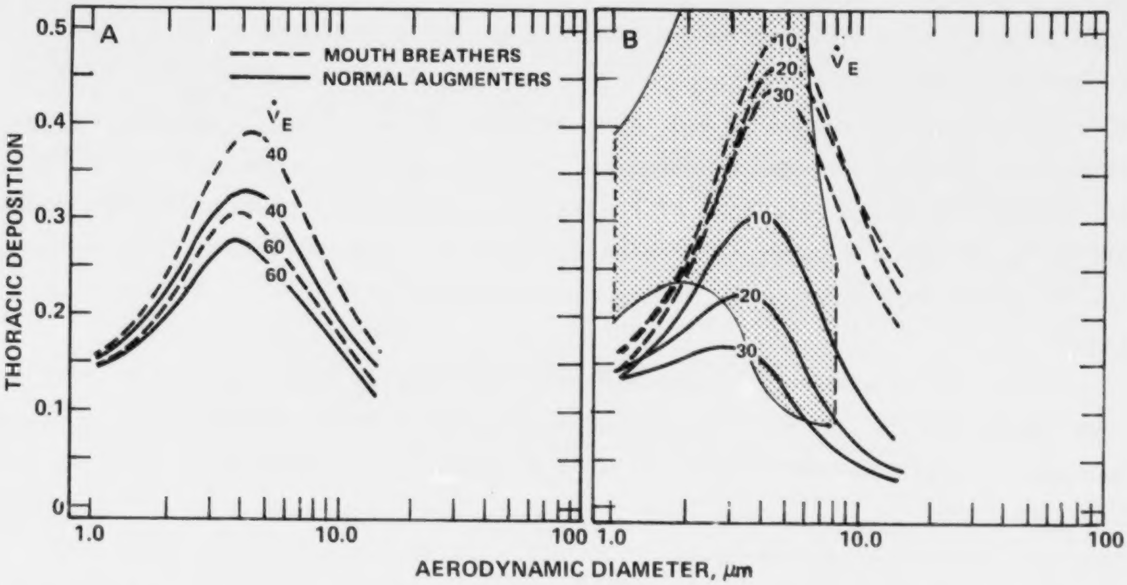

Figure 3. Estimates of thoracic deposition of particles between 1 and 15 μm by Miller et al. (1986) for normal augmenters (solid lines) and mouth breathers (broken lines) are shown for minute ventilation ($\dot{V}_E$) exceeding the switch point of 35 L min^{-1} (A) and for lower $\dot{V}_E$ (B). Normal augmenters are individuals who normally use oronasal breathing to augment respiratory airflow when $\dot{V}_E$ exceeds about 35 L min^{-1}, while mouth breather refers to those individuals who habitually breathe oronasally (Niinimaa et al., 1981). The shaded area (B) is a composite of the computed bands of thoracic deposition of particles less than 8 μm by Swift and Proctor (1982) for $\dot{V}_E$ of approximately 24.6 and 15 L min^{-1}.

scientists dealing primarily with radiological protection issues (Hofmann et al., 1979; Hofmann, 1982a,b; Crawford, 1982). More recently, Phalen et al. (1985) have studied the postnatal enlargement of human tracheobronchial airways and its implication for the deposition of particles ranging from 0.05 to 10 μm in size. They made some morphometric measurements in replica lung casts of people aged 11 days to 21 years. The model predictions for deposition during inspiration only were computed for three states of physical exertion -- low activity, light exertion, and heavy exertion. Scaling techniques were employed to make age-dependent adjustments from adult flow rates.

While the predictions of Phalen et al. (1985) indicate that, in general, increasing age is associated with decreasing particulate deposition efficiency, high flow rates and large particulate sizes do not exhibit consistent age-dependent differences. Since $\dot{V}_E$ at a given state of activity is approximately linearly related to body mass, children will inhale more air per unit body mass, resulting in higher TB doses. For resting ventilation, this age-related dose effect, as a function of particulate size, is illustrated in Figure 4. Xu and Yu (1986) also computed particle deposition efficiencies as a function of age utilizing the growth models of Hoffman et al. (1979, 1982a,b). They take into account the age dependence of head deposition in their calculations. In contrast to the predictions of Phalen, et al (1985), a peak in TB deposition efficency at about 2 years is predicted. However, when divided by body weight, the TB deposition rate would show an age dependence similar to Phalen et al. (1985).

While children may be at greater risk than adults from exposure to particulate matter on the basis of deposition during inspiration, information is needed on possible age-dependent differences in ET deposition, deposition over the entire breathing cycle, mucociliary clearance, and tissue sensitivity, in order to place this risk into perspective relative to health effects evaluations.

Other deposition characteristics of individuals and atmospheric distributions (as well as other factors) can cause variations in regional deposition. The following examples illustrate potentially important variations in exposure/deposition patterns:

(1) The peak in alveolar deposition efficiency for nasal and mouth-only breathing (Figure 2) tends to occur at or near the normal minimum in the bimodal distribution (2 to 4 µm MMAD). However, near emission sources or in other polluted conditions, substantial increases can occur in the coarse- or fine-mode contribution to this most efficiently deposited range.

(2) The deposition of both coarse and fine particles in the tracheobronchial region can be increased over normal ranges by increased breathing rates during exercise and by cigarette smoking, in both bronchitic and asthmatic subjects, generally reducing alveolar deposition. Since retention of particles at 24 hr was significantly lower when bronchoconstriction was induced before inhalation of particles than when bronchoconstriction was induced after inhalation, Svartengren et al. (1984) postulated that bronchoconstriction may serve as a defense mechanism for the alveolar region. However, enhanced tracheobronchial deposition may not be protective, especially for disease states (e.g., bronchitis) or other conditions that constrict, inflame, or cause mucous build-up in airways. Further complicating our understanding of lung clearance mechanisms in obstructive airways disease is the variety of mucociliary transport patterns that can be observed, including regurgitation, stasis, spiral motion, and movement toward the opposite bronchus (Isawa et al., 1984).

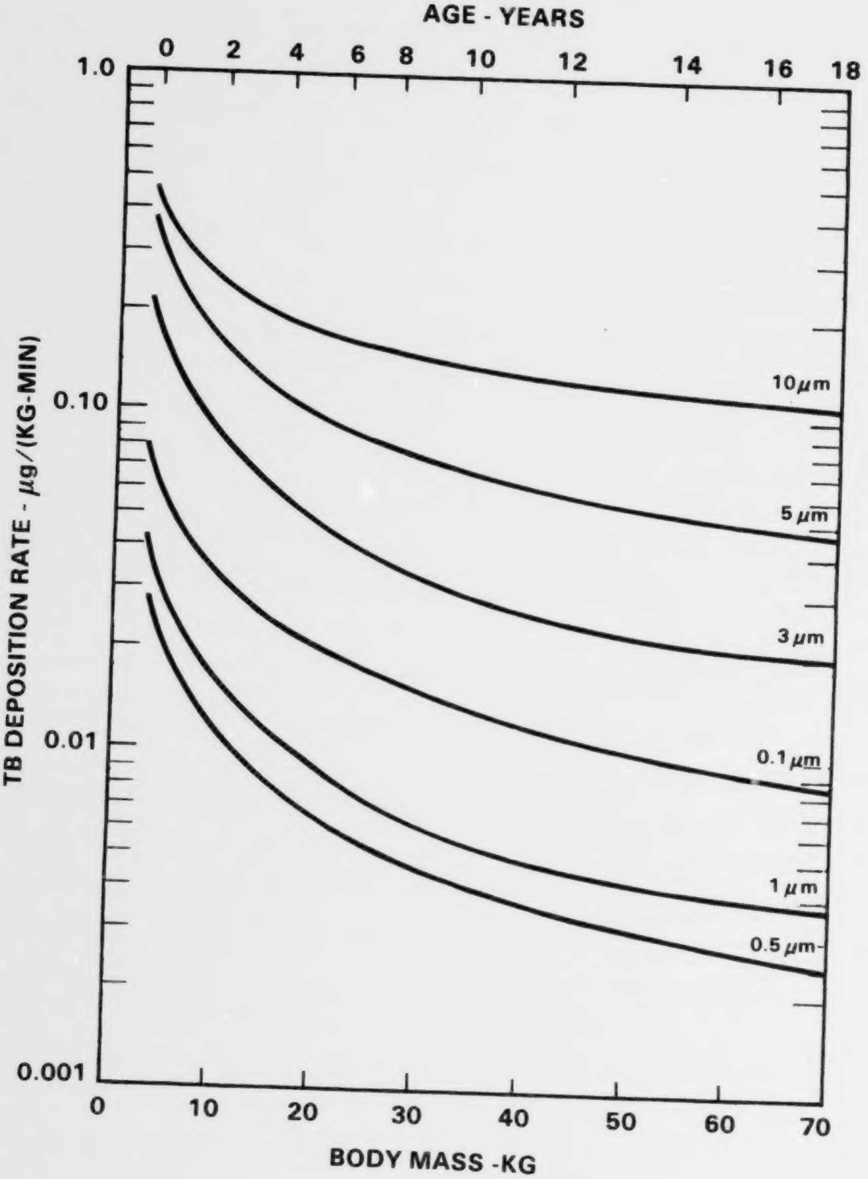

Figure 4. Predicted initial dose to the TB region as a function of body mass. Assumptions include equivalent upper airway deposition for all ages, inhalation of particles at 1 mg/m³ concentration in air, and resting minute ventilation.

Source: From Phalen et al. (1985).

(3) Regional mass deposition data do not provide insights regarding localized "hot spot" deposition. Significantly higher particulate mass to lung surface ratios can occur in the extrathoracic and tracheobronchial regions as compared to the alveolar region. Gerrity et al. (1979) computed the average particle surface concentration of an inhaled 8 μm MMAD aerosols in each generation of the Weibel lung model (Weibel, 1963) and predicted as much as two orders of magnitude difference between particulate surface concentration in the segmental bronchi compared to terminal bronchioles. Local surface concentrations of deposited particles within large airways are probably higher than the average. Also, respiratory disease states that result in altered breathing patterns (e.g., increased oral breathing) may lead to increased deposition of particles in particular respiratory tract regions.

(4) Although the probability of deposition of particles larger than 10 μm in the alveolar region is low, small numbers of such particles have been found in human lungs (U.S. EPA, 1982a,b). Some evidence suggests that those large insoluble coarse substances that do penetrate may be cleared at a much slower rate. Animal tests indicate that 15 μm particles instilled in this region clear much more slowly than smaller particles of the same composition (U.S. EPA, 1982a,b).

Besides variations in regional deposition patterns found for inhaled particles and factors affecting typical deposition patterns, regional differences exist for clearance mechanisms by which inhaled particles penetrating various levels of the respiratory tract are removed. The effects of inhaled particulate matter and other noxious agents, e.g., irritative gases, on clearance mechanisms represent one of the major categories of toxic actions exerted by such air pollutants. Detailed reviews of clearance mechanisms and effects on them due to inhaled particles and sulfur dioxide (SO_2) are presented elsewhere (U.S. EPA, 1982a,b; Lippmann et al., 1982; Lippmann and Schlesinger, 1984).

Mucociliary clearance and alveolar clearance mechanisms are of most concern here. Lung mucociliary clearance is the major defense mechanism by which inhaled particles deposited in the tracheobronchial airways are removed from the respiratory tract. Particle-laden mucus is transported by the tips of cilia which are immersed in an aqueous sol layer. Airway mucus transport rates decrease distally from the trachea (Asmundsson and Kilburn, 1970; Foster et al., 1980) with particle residence times of potentially as much as 300 minutes in the terminal bronchioles (Lee et al., 1979). Mucociliary clearance half-times of the healthy lung can range typically between 30 minutes and several hours, depending on the initial distribution of particles and mucus transport rates within each airway. Lung mucociliary clearance can be impaired by disease states of the lungs (Lippman et al., 1980). Svartengren et al. (1986) have observed marked dysfunction of lung mucociliary clearance in patients with bronchiectasis. Influenza A and respiratory syncitial virus infection cause a decrease in lung mucociliary clearance (Camner et al., 1973; Levandowski et al., 1985; Garrard et al., 1985) and a virtual halt in tracheal mucus transport (Levandoswki et al., 1985) unless supplemented by cough. Retarded mucus transport within the lungs can lead to increased residence times of inhaled particles.

Two general types of alveolar clearance mechanisms are generally recognized: absorptive and non-absorptive. Absorptive mechanisms involve active and passive transport processes, whereby deposited particles permeate the alveolar epithelium and penetration of endothelial barriers occurs prior to uptake into the blood or lymphatic transport. These processes are most effective in removing highly soluble particles. Phagocytosis of deposited particles by alveolar macrophages is generally accepted as the chief non-absorptive clearance process. Some low-solubility materials may escape phagocytosis and accumulate as focal deposits within parenchymal tissues. In the International Commission on Radiological Protection (1979) lung model it has been suggested that as much as 40 percent of particles deposited in alveoli migrate, either free or phagocytized, to the distal portions of the ciliated airways for subsequent removal by mucociliary clearance. Alveolar clearance rates depend in large part on particle solubility. Several studies of long-term clearance of highly insoluble particles in the 1- to 4-μm range (Bailey et al., 1982; Bohning et al., 1982; Philipson et al., 1985) report two phases with half times of approximately 20 and 300 days, though Philipson et al. (1985) observed slow half-times of as much as 2500 days. Stahlhofen et al. (1980) measured the long-term clearance of ferric oxide particles (moderately insoluble) between 1 and 9 μm MMAD and found single phase clearance half-times of between 70 days for the smaller particles and 110 days for the larger ones.

Continuous exposures to ambient aerosols result in the simultaneous deposition and redistribution of particles. The regional dose of particles inhaled continuously may thus differ significantly from the regional pattern of acute aerosol deposition. Brain and Valberg (1974) developed a model of retention of continuously inhaled particles based on the ICRP (Task Group on Lung Dynamics, 1966) lung model. Gerrity et al. (1983) further refined it to the Weibel (1963) lung model, taking into account individual airway mucus transport rates. The Gerrity et al. (1983) model predicts maximum doses to the trachea and respiratory bronchioles for a moderately insoluble 10-μm aerosol.

Deposition of inhaled sulfate compounds in the respiratory tract is complex and depends upon breathing patterns and physical properties of the inhaled particles. Deposition patterns and clearance mechanisms for sulfates depend upon their particular size ranges (mainly fine particles <2.5 μm) as discussed above. Of most importance is the fact that deeper penetration of particles into the respiratory tract occurs during breathing through the mouth or oronasally than during nasal breathing.

Of particular concern from a health standpoint is the fact that acidic aerosols exist in ambient air mainly in the size range of 0.3 to 0.6 μm (MMAD), well within the range of readily inhalable fine-mode particles capable of penetrating deeply into tracheobronchial and alveolar regions of the respiratory tract. Under fog conditions, where acidic components are often incorporated into water droplets of larger sizes up to 10-15 μm, concern exists in regard to the potential for health effects being associated with the increased deposition of acidic fog droplets in the tracheobronchial regions of the respiratory tract.

2.2 SULFUR DIOXIDE UPTAKE AND FATE

As discussed in U.S. EPA (1982a,c), sulfur dioxide is soluble in water and readily absorbed upon contact with the moist surfaces of the nose and upper respiratory passages. It is well established that the gas is almost completely removed (95 to 99 percent) by nasal absorption under resting conditions in both man and laboratory animals. A recent study by Schachter and coworkers (1986) also indicates similar, almost complete, removal of SO_2 in nasal passages during nasal breathing under increased exercise conditions. Schachter et al. (1986) exposed six subjects to 2.62 mg SO_2/m^3 (1 ppm) in an environmental chamber to study nasal absorption of inhaled SO_2. A 6 min rest was followed by 4 to 6 min of exercise at 450 kpm during which subjects breathed only via the nose. A catheter was placed in the oral cavity and connected to an SO_2 analyzer. No detectable quantities of SO_2 could be measured when sampling from the mouth. In addition, saliva samples were analyzed for dissolved SO_2; no dissolved SO_2 was detected. These results confirm previous observations that the nose is extremely efficient in removing SO_2.

Other human studies indicate that SO_2 penetration to the lower respiratory tract increases with activity and increased ventilation associated with a shift from nasal to oronasal breathing at a mean $\dot{V}_E$ of 30 L min^{-1} (Niinimaa et al., 1980, 1981; D'Alfonso, 1980). Most studies on the deposition of SO_2 in animals and humans have been done at concentrations greater than 2.62 mg/m^3 (1 ppm). The 95 to 99 percent removal of SO_2 by the upper respiratory tract has not been confirmed at levels ordinarily found in ambient air (generally less than 0.1 mg/m^3 [0.038 ppm]). It is expected, however, that similar deposition patterns would be observed at these lower concentrations of SO_2. Once inhaled, SO_2 is absorbed quickly into the mucus layer lining the ET and TB regions, where reactions can occur which might result in alterations in the viscosity of mucus. Absorbed SO_2 can also be transferred rapidly into the systemic circulation. Less than 15 percent of the total inhaled SO_2 is likely to be exhaled immediately, with only small amounts (about 3 percent) being desorbed during the first 15 minutes after the end of exposure (U.S. EPA, 1982a,b).

2.3 POTENTIAL MECHANISMS OF TOXICITY ASSOCIATED WITH INHALED PARTICLES AND SO_2

U.S. EPA (1982a) noted that numerous possibilities exist by which a wide variety of toxic effects may be exerted by inhaled particles once deposited in the respiratory tract. Certain general types of mechanisms of toxicity can be identified to apply across a wide range of mixtures of inhaled particles, either acting alone or in combination with other common gaseous air pollutants, such as SO_2, NO_x, or ozone. These include, for example, possible irritant effects that result in decreased air flow due to airway constriction, altered mucociliary transport and effects on alveolar macrophage activity. Other toxic effects and underlying mechanisms of action are much more chemical-specific, and depending on the particular materials involved, may include forms of systemic toxicity involving non-respiratory system organs and functions. The main focus of discussion here is on general mechanisms of toxicity rather than more chemical-specific ones.

The tracheobronchial portion of the respiratory system is the site of deposition of a mixture of fine (especially hygroscopically fine) and relatively small (<10-15 μm) coarse-mode particles. Bronchoconstriction is one common response to deposition of particles in this region and has been reported in response to short-term exposure to high levels of various "inert" dusts, as well as acid and alkaline aerosols of varying particle sizes. Bronchoconstriction produced by acute exposures is likely because of neurologically-mediated reflexive actions arising from chemical and/or mechanical stimulation of irritant neural receptors in the bronchi. Since particle deposition and epithelial nerve endings tend to concentrate near airway bifurcations, deposition at such points may exert an influence on pulmonary mechanical changes due to chemical or mechanical stimulation of receptors. Reflex coughing and bronchoconstriction due to irritant effects of particles or SO_2 on tracheobronchial region receptors may be related to effects observed in various epidemiological studies, e.g., aggravation of chronic respiratory disease states such as asthma, bronchitis, and emphysema. Also, as noted earlier, some persons with asthma or other respiratory diseases may have elevated particle deposition rates in the tracheobronchial region which may contribute to a cascading effect of further bronchoconstriction and increased particle deposition in that region.

Referring to the earlier discussion of particle clearance mechanisms, several more potential mechanisms of toxicity associated with inhalation of airborne particles can be readily discerned. This includes a plausible sequence of events by which inhaled particles can contribute to chronic obstructive pulmonary disease (Albert et al., 1973; Lippmann et al., 1980). That is, inhaled particles and noxious gases can stimulate changes in the distribution and activities of various cell types lining the tracheobronchial airways. Acute exposures to high levels of airborne particles initially stimulate increased mucus secretion and mucociliary flow useful in clearing inhaled particles. However, with continuous or repeated exposures, more marked changes can occur, e.g., marked and persistent depression in bronchial clearance, increase in secretory cell number, increase in the thickness of the mucous layer (Lippman and Schlesinger, 1984). Also, certain particles and gases affect the number of ciliated cells or their functioning so as to alter (i.e., speed or slow) mucociliary clearance rates. Mucociliary clearance is affected by fine sulfuric acid aerosols, high levels of carbon dust, and cigarette smoke.

Because of the above mucociliary clearance phenomena, airborne particles may be importantly involved as etiological factors that contribute to various types of chronic lung diseases, as discussed by U.S. EPA (1982a,b) and Lippmann et al. (1980). This includes: likely involvement in the pathogenesis of chronic bronchitis; increasing susceptibility to acute bacterial and viral infections, especially in populations or groups (e.g., children, the elderly and cigarette smokers) already predisposed to such infections by other factors; and likely aggravation of preexisting disease states, e.g., chronic bronchitis or emphysema, or other respiratory conditions such as bronchial asthma. Also, some individuals (e.g., those with Kartagener's syndrome) have genetically inherited defects in ciliated cell function or other disease states, which result in much reduced mucociliary clearance of inhaled particles and potentially greater vulnerability to toxic effects of such particles.

Particle deposition within the alveolar region of the lungs is mainly limited to fine and coarse particles of less than 10 μm D_{ae}. Several important characteristics in the alveolar region affect responses to inhaled particles. Clearance from the alveolar region is much slower than from the tracheobronchial region. The alveolar region is the site of oxygen uptake and of various non-respiratory functions of the lungs that may be affected by pollutant exposures. Many victims of London air pollution episodes were patients suffering from cardiopulmonary diseases (e.g., emphysema and bronchitis), which normally reduce the lungs' ability to transfer oxygen to blood. Individuals with chronic lung disease and nonuniform ventilation distribution will be sensitive to pollution if only because the delivered dose to the region that is being ventilated will be higher than it would be if ventilation were normally distributed. Although this added load (due to pollution exposure) is usually tolerable in normal individuals, the added stress and chain of events may lead to fatal or irreversible damage in individuals already compromised with cardiopulmonary disease.

2.4 SUMMARY AND CONCLUSIONS

Studies published since preparation of the earlier criteria document (U.S. EPA, 1982a) and the previous addendum (U.S. EPA, 1982c) support the conclusions reached at that time and provide clarification of several issues. In light of previously available data, new literature was reviewed with a focus towards (1) the thoracic deposition and clearance of large particles, (2) assessment of deposition during oronasal breathing, (3) deposition in possibly susceptible subpopulations, such as children, and (4) information that would relate the data to refinement or interpretation of ancillary issues, such as inter- and intrasubject variability in deposition, deposition of monodisperse versus polydisperse aerosols, etc. Major results for the first three areas are given below.

The thoracic deposition of particles ≥ 10 µm D_{ae} and their distribution in the TB and P regions was studied by a number of investigators (Svartengren, 1986; Heyder, 1986; Emmett et al., 1982). Depending upon the breathing regimen used, TB deposition ranged from 0.14 to 0.36 for 10-µm D_{ae} particles, while the range for 12-µm D_{ae} particles was 0.09 to 0.27. For particles 16.4 µm D_{ae}, a maximally deep inhalation pattern resulted in TB deposition of 0.12.

The experimental data cited above were obtained from human exposure studies in which the subjects inhaled through a mouthpiece. Some of the minute ventilations employed would more normally occur with oronasal breathing (partly via the mouth and partly nasally). Various studies (Swift and Proctor, 1982; Miller et al., 1984, 1986) have simulated deposition during oronasal breathing by adjusting for parallel nasal and oral deposition as a function of air flow through the respective compartments. While the magnitude of deposition in various regions depends heavily upon minute ventilation, there is, in general, a gradual decline in thoracic deposition for large particle sizes, and there can be significant deposition of particles greater than 10 µm D_{ae}, particularly for individuals who habitually breathe through their mouth. Thus, the deposition experiments wherein subjects inhale through a mouthpiece are relevant to examining the potential of particles to penetrate to the lower respiratory tract and pose a potentially increased risk. Increased risk may be due to increased localized dose of larger particles (Gerrity et al., 1983).

Although experimental data are not currently available for deposition of particles in the lungs of children, some trends are evident from the modeling results of Phalen et al. (1985). Phalen and co-workers made morphometric measurements in replica lung casts of people aged 11 days to 21 years and modeled deposition during inspiration as a function of activity level. They found that, in general, increasing age is associated with decreasing particulate deposition efficiency. However, very high flow rates and large particulate sizes do not exhibit consistent age-dependent differences. Xu and Yu

(1986) found a small peak in deposition efficiency at 2 years of age followed by a decline with increasing age. When divided by body weight, though, their results qualitatively agree with those of Phalen et al. (1985). Since minute ventilation at a given state of activity is approximately linearly related to body mass, children receive a higher TB dose of particles than do adults and would appear to be at a greater risk, other factors (i.e., mucociliary clearance, particulate losses in the head, tissue sensitivity, etc.) being equal.

CHAPTER 3. EPIDEMIOLOGICAL STUDIES OF HEALTH EFFECTS ASSOCIATED WITH EXPOSURE TO AIRBORNE PARTICLES AND/OR SULFUR OXIDES

Extensive published information exists concerning health effects associated with exposure to airborne particulate matter and sulfur oxides. Detailed evaluations of much of this extensive literature (including discussions of potential mechanisms of toxicity and findings emerging from animal toxicology experiments, controlled human exposure studies, and epidemiological studies) are provided in the 1982 EPA criteria document (U.S. EPA, 1982a), as well as several other critical reviews of the subject (World Health Organization, 1979; Holland et al., 1979; Lippmann et al., 1980; Lippmann and Schlesinger, 1984). Key health effects findings emerging from the earlier criteria review (U.S. EPA, 1982a) are summarized below, providing a perspective against which more recently published studies are then highlighted and evaluated in the present chapter.

3.1. HUMAN HEALTH EFFECTS ASSOCIATED WITH SHORT-TERM EXPOSURES

As reviewed by U.S. EPA (1982a), much information has been generated by experimental animal studies and controlled human exposure studies in regard to health effects associated with short-term (<24 hr.) exposures to airborne particles and sulfur oxides. Especially important information concerning the effects of acute sulfur dioxide exposures on pulmonary functions has been derived from controlled human exposure studies, as later discussed in Chapter 4 of this Addendum. However, other crucial information gained in regard to effects on human health of short-term exposures to realistic concentrations of sulfur oxides and/or airborne particles has come from epidemiological studies. Complicating such studies are the frequent co-occurrence of elevated levels of sulfur oxides along with airborne particles and difficulties in adequately controlling or adjusting for the effects of other potentially confounding variables. Attention is directed here mainly to identification of epidemiological studies that yield information relevant to the delineation of exposure-effect or exposure-response relationships.

3.1.1. Mortality Effects of Short-Term Exposures

As discussed in U.S. EPA (1982a), the most clearly defined effects on mortality arising from exposure to sulfur oxides and particulate matter have been sudden increases in the number of deaths occurring, on a day-to-day basis, during episodes of high pollution. The most notable of these occurred in the Meuse Valley in 1930, in Donora in 1948, and in London in 1952. Additional episodes with notable increases in mortality occurred in London during various winters from 1948 to 1962. Besides evaluating mortality associated with major episodes, epidemiology studies also focused on more moderate day-to-day variations in mortality within large cities in relation to PM and SO_x pollution.

The large body of literature concerning such studies carried out in the United Kingdom, elsewhere in Europe, the United States and Japan was critically reviewed in detail by U.S. EPA (1982a). As discussed there, various methodological problems with most of the studies precluded drawing of quantitative conclusions regarding exposure-effect or exposure-response relationships of importance for deriving air quality standards. Among the main problems were inadequate measurement or control for potentially confounding variables and inadequate quantitation of exposure to airborne particles, SO_2 or other associated pollutants (e.g., sulfates).

Despite such problems, U.S. EPA (1982a) concluded that the then available studies collectively indicated that mortality was clearly and substantially increased when airborne particle 24-hr concentrations exceeded 1000 $\mu g/m^3$ (as measured by the BS method) in conjunction with elevations of SO_2 levels in excess of 1000 $\mu g/m^3$ (with the elderly or others with severe preexisting cardiovascular or respiratory disease mainly being affected). As for evaluation of risks of mortality at lower exposure levels, U.S. EPA (1982a) concluded that studies conducted in London by Martin and Bradley (1960) and Martin (1964) yielded useful, credible bases by which to derive conclusions concerning quantitative exposure-effect relationships. Table 1 summarizes key conclusions drawn from these and other critical studies of mortality and morbidity effects associated with short-term (24-hr) exposures to particulate matter and SO_2, as stated earlier in the 1982 EPA criteria document (U.S. EPA 1982a).

TABLE 1. SUMMARY OF QUANTITATIVE CONCLUSIONS DRAWN IN U.S. EPA (1982a) FROM EPIDEMIOLOGICAL STUDIES RELATING HEALTH EFFECTS TO ACUTE EXPOSURE TO AMBIENT AIR LEVELS OF SO_2 AND PM IN LONDON**

Type of Study	Effects observed	24-hr average pollutant level (μg/m^3)		Reference
		BS	SO_2	
Mortality	Clear increases in daily total mortality or excess mortality above a 15-day moving average among the elderly and persons with preexisting respiratory or cardiac disease during the London winter of 1958-59.	≥1000	≥1000	Martin and Bradley (1960); Martin (1964)
	Analogous increases in daily mortality in London during 1958-59 to 1971-72 winters.			Mazumdar et al. (1981)
	Some indications of likely increases in daily total mortality during the 1958-59 London winter, with greatest certainty (95% confidence) of increases occurring at BS and SO_2 levels above 750 μg/m^3.	500-1000	500-1000	Martin and Bradley (1960)
	Analogous indications of increased mortality during 1958-59 to 1971-72 London winters, again with greatest certainty at BS and SO_2 levels above 750 μg/m^3 but indications of small increases at BS levels <500 μg/m^3 and possibly as low as 150-200 μg/m^3.			Mazumdar et al. (1981)
Morbidity	Worsening of health status among a group of chronic bronchitis patients in London during winters from 1955 to 1960.	≥250-500*	≥500-600	Lawther (1958); Lawther et al. (1970)
	No detectable effects in most bronchitics; but positive associations between worsening of health status among a selected group of highly sensitive chronic bronchitis patients and London BS and SO_2 levels during 1967-68 winter.	<250*	<500	Lawther et al. (1970)

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*Note that the 250-500 μg/m^3 BS levels stated here may represent somewhat higher PM concentrations than those actually associated with the observed effects reported by Lawther (1970). This is due to the estimates of PM mass (in μg/m^3 BS) used by Lawther being based on the D.S.I.R. calibration curve found by Waller (1964) to approximate closely a site-specific calibration curve developed by Waller in central London in 1956, but yielding somewhat higher mass estimates than another site-specific calibration developed by Waller a short distance away in 1963. However, the precise relationship between estimated BS mass values based on the D.S.I.R. curve versus the 1963 Waller curve cannot be clearly determined due to several factors, including the non-linearity of the two curves and their convergence at low BS reflectance values.

*Source: U.S. EPA (1982a). Subsequent reanalyses of the London mortality data alluded to here have been carried out since completion of U.S. EPA (1982a) and are described elsewhere in this Chapter. In general, the results of these more recent reanalyses demonstrate relatively continuous exposure-response relationships across the entire range of BS levels reported for London during the winters of 1958-59 to 1971-72, with no clear thresholds evident for significant associations between daily mortality and BS (but not SO_2) at levels ranging to below 250-500 μg/m^3. The difference in the gravimetric calibrations noted above for 1956 and 1963 and lack of later gravimetric calibration of BS readings, however, limit specification of precise PM levels (in μg/m^3) associated with the relatively small increases in mortality seen at lower BS concentrations. In addition, new morbidity studies regarding effects of short-term PM/SO_x exposures of a more contemporary nature are also discussed elsewhere in this chapter.

The studies by Martin and Bradley (1960) and Martin (1964) dealt with a relatively small body of data on relationships between daily mortality in Greater London and daily variations in pollution (smoke and sulfur dioxide) during the winter of 1958-59. Aerometric data from multiple sampling sites used in their analysis can be considered reasonably representative of outdoor concentrations in the areas where people lived, although the inclusion of outer, less-densely populated areas meant that average exposure may have been underestimated. During the winter of 1958-59, Martin and Bradley (1960) reported that mortality increased on some days when smoke concentrations increased by more than 100 $\mu g/m^3$ over the previous day or when SO_2 concentrations increased by 70 $\mu g/m^3$ (0.025 ppm). Increases in daily mortality were up to about 1.2 times expected values assessed from 15-day moving averages. Thick fog (visibility less than 200 meters) was also associated with increases in mortality. The relative importance of the three factors (smoke, SO_2, fog) could not be clearly determined, but on the basis of other work, the authors considered that smoke was probably most important. When results were considered on an absolute basis (Lawther, 1963), it was concluded that increases in mortality became evident when the 24-hr mean concentrations of smoke and sulfur dioxide exceeded 750 $\mu g/m^3$ and 710 $\mu g/m^3$ (~0.25 ppm), respectively. Studies on day-to-day variations in mortality in London were continued in successive winters and coupled with the records of emergency hospital admissions. Martin (1964) showed correlations between both the daily mortality and hospital admission data and concentrations of smoke or SO_2. There was no clearly defined level (threshold) above which effects were seen, but fairly consistent increases in both mortality and hospital admissions occurred when concentrations of smoke and sulfur dioxide each exceeded a 24-hr mean of about 500 $\mu g/m^3$. Based on the above analyses and a reanalysis of the Martin and Bradley data set by Ware et al. (1981), U.S. EPA (1982a) concluded that notable increases in mortality among the elderly and chronically ill may have been associated with BS and SO_2 levels in the range of 500 to 1000 $\mu g/m^3$. Much less certainty was attached to suggestions of possible slight increases in mortality at still lower BS or SO_2 concentrations, based on the Ware et al. (1981) reanalyses.

In subsequent years, because of reductions in London BS levels brought about by implementation of the British Clean Air Act and more gradual SO_2 reductions, only few occasions occurred when smoke or SO_2 levels exceeded 500

$\mu g/m^3$. Analyses of daily mortality in London in relation to variations in smoke and SO_2 levels during winters from 1958-59 to 1971-72 were reported by Mazumdar et al. (1981). These analyses are of special value in attempting to define lowest levels of exposure to particulate matter and/or SO_2 associated with increased mortality, because they include winters when levels of those pollutants never exceeded 500 $\mu g/m^3$. The results obtained for airborne particles (measured in terms of BS) were analyzed in relation to linear and quadratic models, which Mazumdar et al. (1981) found to provide good fits to the data examined after relevant potentially confounding variables, e.g., temperature and humidity, were taken into account statistically. U.S. EPA (1982a) concluded that both of tne models suggest small increases in mortality at smoke levels below 500 $\mu g/m^3$ and, possibly, to as low as 150-250 $\mu g/m^3$.

In a publication newly available since completion of U.S. EPA (1982a), Mazumdar et al. (1982) reported further on three types of analyses of London mortality during the 1958-59 to 1971-72 winters: (1) year-by-year multiple regressions, (2) stratification using nested quartiles of one pollutant within another, and (3) multiple regression of a subset of high-pollution days. Steps were taken in each analysis to control for potentially confounding factors. Mortality and pollution variables were first divided by their winter means (indexed or percent) to adjust for year-to-year variation. Seasonal trends were adjusted for by treating each variable as a deviation (residual) from 15-day moving averages; these residuals were then corrected for weather factors by regressing separately indexed mortality, SO_2 and smoke residuals in temperature and humidity residuals of the same day, previous day and lag days up to 1 wk; and dummy variables were used to remove day-of-week effects. The corrected indexed pollution variables were then reconverted to absolute units by multiplying each value by the corresponding winter mean, but the mortality values were left in indexed form.

Mazumdar et al. (1982) reported that the year-by-year multiple regressions yielded generally much smaller coefficients for SO_2 (14 winter $\bar{x} = 1.17$ percent mortality increase/mg/m^3 SO_2; $p > 0.10$) versus those for smoke (14 winter $\bar{x} =$ 25.09 percent/mg/m^3 smoke; $p < 0.01$). Also, the nested quartile analyses using 16 cells (i.e., 4 quartiles of smoke within 4 quartiles of SO_2 and vice versa), were reported as only partially successful, in that substantial covariation remained between the two pollutants in the highest and lowest quartiles. Visual inspection of other cells, the authors noted, nevertheless suggested a much larger smoke than SO_2 effect. Last, multiple regression analyses, using the 100 days during the 14 winters when the two pollutants were in their highest deciles (excluding 5 days during the 1962 episode), were reported as showing that mortality increases monotonically with smoke for fixed SO_2 levels but mortality only increased with SO_2 levels above 0.7 mg/m^3 for fixed smoke levels. The authors concluded that their analyses of London data for 14 winters support the conclusion that mortality was significantly and positively associated with air pollution, but the mortality/pollution association was almost entirely due to smoke. They also noted possible contributions of SO_2 at sufficiently high pollutant levels (i.e., when both SO_2 and smoke >0.7 mg/m^3). Results from linear and quadratic models of mortality regressed on smoke alone led the authors to state a preference for the quadratic model supplemented by a hypothesis that at low smoke levels (<0.3 mg/m^3), smoke may serve as a surrogate for an unidentified variable (e.g., a highly toxic fraction of particulate emissions).

More recently, Ostro (1984) reported that new analyses of the same 1958-59 to 1971-72 London winter data indicate some risk of mortality even at smoke levels below 150 $\mu g/m^3$. Specifically, Ostro (1984) employed a variation of a standard multiple regression model to test whether the data supported the existence of a "threshold" at BS = 150 $\mu g/m^3$. Observations across the range of pollutant levels were divided into two segments, those falling below versus those above 150 $\mu g/m^3$. Regression analyses for data below 150 $\mu g/m^3$, controlling for important potentially confounding factors (e.g., temperature, humidity, etc.), indicated a statistically significant pollutant effect on mortality below the BS = 150 $\mu g/m^3$ level. For 11 of 14 winters, the coefficients for mortality associations with BS values below 150 were statistically different from zero at $p \leq 0.10$. Additional analyses focused on the last seven winters, starting in 1965-66, during which there were no BS values above 500 $\mu g/m^3$. The mortality coefficients were significant at $p < 0.05$ for six years and at the 0.01 level in four of the years. Ostro (1984) concluded that these results are suggestive of a strong association of BS with mortality, holding temperature and humidity constant, at levels below 150 $\mu g/m^3$.

The Mazumdar et al. (1982) and Ostro (1984) analyses produced generally analogous results in relation to reported findings on PM effects: (1) each found significant positive associations between increased mortality and BS levels for most of the 14 London winters from 1958-59 to 1970-71, when the data were analyzed on a year-by-year basis; (2) the coefficients obtained for mortality associations with lower BS values were generally larger than values obtained with higher BS levels, an apparently counterintuitive result; and (3) no clearly defined threshold for BS-mortality associations could be identified based on either set of analyses, both of which showed small but significant associations at levels below 500 $\mu g/m^3$ BS.

No readily obvious reasons stand out as explaining the reported stronger correlations between lower BS values and mortality than associations seen at higher BS levels, although both Mazumdar et al. (1982) and Ostro (1984) tendered some possibilities (for example, the low levels of smoke in later years may have contained higher proportions of respirable particles or specific toxic materials). Still other questions have been raised in regard to these analyses; for example: (1) whether or not the effects of smoke and SO_2 can be credibly separated out, given the very high correlation (generally ≥ 0.80 or 0.90) between BS and SO_2 levels in the subject data set; (2) whether unmeasured variables, such as indoor air pollution levels, might have also covaried with outdoor BS and SO_2 concentrations and contributed to observed mortality effects; or (3) whether other unevaluated longer-term changes in demographic characteristics of the London population (age, socioeconomic levels, ethnic mix, etc.) over the 14 winters might not be such as to contribute to spurious apparent associations between mortality increases and BS or SO_2. Also, Roth et al. (1986) suggested that use of deviations of mortality from 15-day moving averages may hide the true relationship between pollution and mortality.

Not all of these issues can be definitively resolved at this time. However, it is unlikely that long-term demographic shifts during the 14 year study period could account for significant year-by-year associations; nor is it likely that indoor air exposures would be consistent from year to year, given variations in yearly climatic conditions coupled with gradual changes in heating practices (shifts away from open hearth burning of coal in residences) that occurred during the 14 year study period. In addition, further reanalyses of the 1958-59 to 1971-72 London mortality data have been carried out in an effort to address issues of the above types. For example, an unpublished analysis of the 1958-59 to 1971-72 London winter data set carried out by Shumway et al. (1983) for the California Air Resources Board (CARB) also produced results indicative of risk below the 500 $\mu g/m^3$ level of smoke. These analyses used a spectral transform multiple regression model and detrending of data to correct for temperature and autocorrelation effects. The best model for predicting cardiovascular, respiratory or overall mortality used lagged temperature and logs of same day levels of SO_2 or smoke. Results were reported to indicate that pollution acts positively and instantaneously, whereas temperature has both a significant same-day effect and a strong negative effect with a lag of two days. The largest portion of variance in daily mortality was attributed to cyclical pollution-temperature patterns typified by 7-21 day periods. Overall, these results suggest that although relatively small increases in PM air pollution may be associated with increased daily mortality in London, the effects were likely greater when higher PM concentrations occurred as part of multi-day cycles than with short duration episodes.

More recent reanalyses, performed by Marcus and Schwartz in cooperation with CARB, are concisely described in Appendix A to this Addendum. The memorandum in Appendix A summarizes their reanalyses as described in an attached more extensive paper (Schwartz and Marcus, 1986) now being prepared for submittal for publication. Their reanalyses indicate that: (1) Clear exposure-response relationships are evident between the main air pollution variables (BS, SO_2 levels) and increases in daily mortality when graphically displayed either in terms of absolute daily mortality or deviations from 15-day moving averages of daily mortality; (2) Multiple regression analyses revealed that either daily mortality or derivations in daily mortality from 15-day moving averages were positively and significantly correlated with increases in BS or SO_2 across the 14 winters, adjusted for time series autocorrelation, temperature, and humidity; (3) Analyses on a year-by-year basis yielded significant linear correlations of mortality with BS for 13 of the 14 winters, including later years only having days $<250\ \mu g/m^3$ and even for 6 of 11 winters when only days with BS $<200\ \mu g/m^3$ were included in the analyses; (4) The partial regression coefficients for BS versus mortality are relatively stable from year to year, although they tend to increase for later years versus the first 7 years; (5) The partial regression relationship between mortality and BS is non-linear, the relationship being convex with somewhat steeper linear slope at lower BS levels ($<250\ \mu g/m^3$) than for higher BS levels ($\geq 500\ \mu g/m^3$); (6) SO_2 is significantly associated with daily mortality, mainly at high levels ($>500\ \mu g/m^3$); but (7) SO_2 effects appear to be somewhat distinguishable from BS effects, with the mortality effects of BS remaining significant and relatively large when SO_2 is included in the regression model whereas inclusion of BS in the model reduces the SO_2 coefficients to insignificant values. Overall, these reanalyses further substantiate and reinforce major results derived from earlier published analyses and point more strongly toward PM-mortality associations even at levels below 150-250 $\mu g/m^3$. On the other hand, it is difficult to estimate with any precision what PM levels (in $\mu g/m^3$) may have been associated with increased mortality at lower BS levels (<150 to 250 $\mu g/m^3$), given lack of contemporaneous gravimetric calibration data beyond 1963.

Taking into account the above considerations, the following conclusions appear to be warranted based on the earlier criteria review (U.S. EPA, 1982a) and present evaluation of newly available analyses of the London mortality experience: (1) Markedly increased mortality occurred, mainly among the elderly and chronically ill, in association with BS and SO_2 concentrations above 1000 $\mu g/m^3$, especially during episodes when such pollutant elevations occurred for several consecutive days; (2) During such episodes coincident high humidity or fog was also likely important, possibly by providing conditions leading to formation of H_2SO_4 or other acidic aerosols; (3) Increased risk of mortality is associated with exposure to BS and SO_2 levels in the range of 500 to 1000 $\mu g/m^3$, for SO_2 most clearly at concentrations in excess of ~700 $\mu g/m^3$; and (4) Convincing evidence indicates that relatively small but statistically significant increases in the risk of mortality exist at BS (but not SO_2) levels below 500 $\mu g/m^3$, with no indications of any specific threshold level having been demonstrated at lower concentrations of BS (e.g., at $\leq$150 $\mu g/m^3$). However, precise quantitative specification of the lower PM levels associated with mortality is not possible, nor can one rule out potential contributions of other possible confounding variables at these low PM levels.

In another study of air pollution relationships with mortality reported since the earlier criteria review (U.S. EPA, 1982a), Mazumdar and Sussman (1983) evaluated associations between mortality events and daily particulate matter and SO_2 levels in Pittsburgh, PA. The analysis, limited to investigation of same-day events, reported a possible relationship between heart disease mortality/morbidity and same day particulate levels (measured in terms of COH), but not same-day SO_2 levels. The analyses specifically evaluated daily mortality rates during 1972-1977 for all of Allegheny County, PA in relation to daily average COH and SO_2 measurements obtained at each of three air monitoring stations: one at the center of the County within a high pollution section of Pittsburgh; another situated relatively near the first in a somewhat less polluted area; and a third in a distinctly cleaner area on the northeast edge of the County. Corrections for trend and seasonal factors were made by use of daily deviations from 15-day moving averages for air pollution, temperature and mortality variables. Multiple regression analyses revealed no statistically significant associations between mortality for all ages or heart disease mortality in relation to either SO_2 or COH when regressed on each variable alone. When SO_2 and COH were considered jointly, only the associations between total or heart disease mortality and COH measurements at the Hazelwood (high pollution area) station were significant at $p < 0.05$. These results, however, cannot be accepted as providing meaningful information on mortality-air pollution associations in the Pittsburgh area in view of: (1) inadequate characterization of county-wide air pollution levels against which to compare mortality rates for the entirety of Allegheny County, the SO_2 and COH levels at each of the three monitoring stations used not being highly correlated (mostly $r \leq 0.4$ to 0.5) with values at the other stations; (2) internal inconsistencies whereby larger coefficients were obtained for associations of mortality to COH readings at the cleaner air station on the edge of the County than the intermediate pollution station near the center of the County; and (3) the use of a large number of separate mortality regression analyses, from among which only two were significant at $p < 0.05$.

In addition to the above reanalyses of London mortality data, reanalyses of mortality data from New York City in relation to air pollution have been recently reported by Ozkaynak and Spengler (1985). These investigators carried out time-series analyses on a subset of New York City data included in a prior analysis by Schimmel (1978) which was critiqued during the earlier criteria review (U.S. EPA, 1982a). The present reanalyses by Ozkaynak and Spengler (1985) evaluated 14 years (1963-76) of daily measurements of mortality (the sum of heart, other circulatory, respiratory, and cancer mortality), COH, SO_2, and temperature. Prior to regression analysis, efforts were made to remove assumed low-frequency confounding by "filtering" each variable to remove its slow-moving components. This included not only use of residuals from 15-day moving averages, but also evaluation of sensitivity of results to other filters. Initial exploratory analyses estimated regression coefficients for COH and SO_2

after all variables were preprocessed with one of several filters (e.g., taking residuals from 7-, 15-, or 21-day moving averages and other filters that removed all cycles in the data that fell beyond indicated periods measured in days). Overall, the regression coefficients for COH ranged from 1.2 to 5.4 daily deaths per unit of COH, most being statistically significant ($p \leq 0.05$). Also, a reasonable range of variation in temperature specifications produced coefficients ranging from 1.3 to 1.8 deaths per COH unit. The risk coefficients of Schimmel (1978) were near the lower end of the range of coefficients found by Ozkynak and Spengler (1985). The latter investigators noted then that they were able to generate a fairly consistent set of estimates by performing a number of sensitivity analyses. They also correctly note that these initial estimates were subject to several technical limitations: (1) misclassification of population exposure can occur in using aerometric data from one fixed monitoring site; (2) the exposure index, COH, is imperfectly related to respirable particle mass levels; and (3) the range of exploratory models initially fit may not have been diverse enough. Consequently, an additional reanalysis was undertaken.

Specifically, more recent reanalysis of the New York City data reported by Ozkaynak and Spengler (1985) used standard time-series methods to control for covariates such as temperature and to handle the problem of autocorrelation. Their previous analysis was also extended by adding records of visibility and weather from three New York City airports, in order to examine spatial homogeneity of daily air pollution in New York City and to use visibility as a surrogate for aerosol extinction (b_{ext}) or for fine particle (FP) pollution as discussed by Ozkaynak et al. (1985). The most salient feature of the mortality data found by this reanalysis was a strong seasonal component which confounds direct regressions involving mortality, air pollution and weather variables. A simple trigonometric expression was used that removed the temperature cyclic component and rendered nonseasonal temperature nonsignificant. Another stationary autoregressive term was also used to exhaust the time-series structure of the mortality records. Consideration of lagged regressions and interactions did not improve the model's predictive ability. Time-series analyses were then performed with a linear model and in a multivariate manner in which corrections for seasonality and autocorrelation were introduced into the linear model. Preliminary estimates of excess deaths (e_i) or elasticities for the pollutant variables were thereby calculated, resulting in the following findings: (1) the time-series analysis showed SO_2 levels to be significantly correlated with mortality (e_{SO2} = 2.3 percent); (2) COH also contributed significantly to excess deaths (e_{COH} = 2.4 percent); (3) B_{ext}, a variable used as a surrogate for FP pollution was also a significant contributor to excess daily deaths ($\underset{\sim}{\ }$1.2 percent); and (4) the total estimated excess deaths attributable to air pollution was $\underset{\sim}{\ }$6.0 percent. The authors concluded that although these are interim results (they are also analyzing the data one year at a time and by each quarter), these findings: (1) indicate that during the study period ambient air pollution of a large urban area was contributing to mortality, (2) appear to corroborate results from cross-sectional mortality studies, and (3) indicate that particulate air pollution, even at current levels, could be of concern for public health. However, the authors again correctly noted limitations of their analyses which preclude full reliance on these preliminary results for risk assessment purposes: (1) The results reflect aggregate analyses of 14 years of data and more thorough analyses need to be done to take into account changing SO_2 and aerosol composition over the period (preliminary analyses indicate no differences in pollutant coefficients for 1963 to 1970 and 1971 to 1976); (2) The results are based on aerometric data from one monitoring station and visibility data from one airport (JFK); and (3) The effects of heat waves and influenza epidemics during the study period have not been considered in any detail in these preliminary analyses.

Hatzakis et al. (1986) recently published a study of short-term effects of air pollution on mortality in Athens, Greece, during 1975-82. Daily concentrations of SO_2 (acidimetric method) and smoke (standard British Method) measured by a five-station network in Athens were evaluated in relation to mortality data abstracted from the Joint Registries of Athens and 18 other contiguous towns in the Greater Athens area. The authors reported that adjusted daily mortality (estimated by subtracting the observed mortality value from an "expected" value, calculated after fitting a sinusoidal curve to the empirical mortality data) was significantly and positively related to SO_2 levels ($b = +0.0058$, $p = 0.05$), but not to smoke levels. Separate multiple regression analyses were done for SO_2 and smoke, controlling in each case for temperature, relative humidity, secular, seasonal, monthly and weekly variations in mortality as well as interactions of the above variables with season. Evaluation of a possible threshold for the SO_2-mortality effect was carried out by successively deleting from the regression model days with the highest SO_2

values. These analyses resulted in the authors suggesting that, if there is an SO_2 threshold, it must lie slightly below 150 μg/m^3 (mean daily value).

The latter result, as stated by the authors, is not consistent with results of other studies in which SO_2 mortality thresholds have been placed around the value of 300 μg/m^3 (or, more credibly, around 500 μg/m^3, as per U.S. EPA, 1982a). Nor is the failure to find significant associations between mortality and smoke consistent with other more usual published findings (although differences in chemical composition of PM in Athens and lack of calibration of smoke readings against gravimetric measurements make it difficult to compare smoke levels from Athens versus elsewhere). Other questions also arise which make it difficult to fully accept the reported findings, e.g.: (1) how representative are the aerometric data for the entire Athens metropolitan area from which the mortality data were abstracted, although the typography of the area, with Athens and adjoining towns situated in a coastal "bowl" surrounded by mountains, and high correlations (mostly r >0.50-0.60) between pollutant readings from the five network stations suggest that the aerometric data may well be quite representative; (2) whether use of deviations of observed mortality data for 1975-82 from expected values derived from 1956-58 mortality data as a pre-high pollution baseline period is statistically sound; (3) whether separate regression analyses for SO_2 and smoke alone are sufficient versus analyses with both these pollutants included; and (4) whether effects of temperature or flu epidemics were adequately compensated for in the analyses.

In summary, the above newly available reanalyses of New York City data raise possibilities that, with additional work, further insights may emerge regarding mortality-air pollution relationships in a large U.S. urban area. However, the interim results reported thus far do not now permit definitive determination of their usefulness for defining exposure-effect relationships, given the above-noted types of caveats and limitations. Similarly, it is presently difficult to accept the findings of mortality associated with relatively low levels of SO_2 pollution in Athens, given questions stated above regarding representativeness of the monitoring data and the statistical soundness of using deviations of mortality from an earlier baseline relatively distant in time. Lastly, the newly reported analyses of mortality-air pollution relationships in Pittsburgh (Allegheny County, PA) utilized inadequate exposure characterization and the results contain sufficient internal inconsistencies, so that the analyses are not useful for delineating mortality relationships with either SO_2 or PM.

3.1.2. Morbidity Effects of Short-Term Exposures

As noted by the World Health Organization (1979), epidemiological studies can be useful in assessing morbidity effects associated with air pollution in different communities or in areas where changes in air pollution occurred over time. In such studies, where respiratory diseases are followed, it is necessary to control for age distribution, socioeconomic status, and other possibly confounding factors. It is also crucial that adequate characterization of exposure to air pollutants of interest be carried out, if quantitative conclusions are to be drawn regarding exposure-effect or dose-response relationships. However, very few of the available epidemiological studies on morbidity effects associated with short-term exposure to airborne particles allow for such conclusions, as evaluated by U.S. EPA (1982a).

Those reported by Lawther for London populations (see Table 1) were identified by U.S. EPA (1982a) as providing credible bases for drawing quantitative-type conclusions about morbidity effects associated with airborne particles (measured as smoke) and elevated SO_2 levels. Lawther et al. (1970) reported on studies carried out from 1954 to 1968 mainly in London, using a diary technique for self-assessment of day-to-day changes in conditions among bronchitic patients. A daily illness score was calculated from the diary data and related to BS and SO_2 levels and weather variables. Pollution data for most of the London studies were mean values from the group of sites used in the mortality/morbidity studies of Martin (1964); those aerometric measurements likely provide reasonable estimates of average exposure in areas where study subjects lived or worked. In early years of the studies, when pollution levels were generally high, well defined peaks in illness score were seen when concentrations of either BS or SO_2 exceeded 1000 $\mu g/m^3$. With later reductions in pollution, the changes in condition became less frequent and of smaller size. From the series of studies as a whole, up to 1968, it was concluded that the minimum pollution levels associated with significant changes in the condition of the patients was a 24-hr mean BS level of ~250 $\mu g/m^3$ together with a 24-hr mean SO_2 concentration of ~500 $\mu g/m^3$ (0.18 ppm). A later study reported by Waller (1971) showed that, with much reduced average levels of pollution, there was an almost complete disappearance of days with smoke levels exceeding 250

$\mu g/m^3$ and SO_2 levels over 500 $\mu g/m^3$ (0.18 ppm). As earlier, some correlation remained between changes in the conditions of the patients and daily concentrations of smoke and SO_2, but the changes were small at these levels and it was difficult to discriminate between pollution effects and those of adverse weather. Thus, as concluded by U.S. EPA (1982a), the observed effects (worsening of health status among chronic bronchitic patients) were clearly associated with BS levels of 250 to 500 $\mu g/m^3$ and, possibly, somewhat lower levels (<250 $\mu g/m^3$) for highly sensitive bronchitic patients.*

Since preparation of U.S. EPA (1982a) evaluations summarized in Table 1, additional studies have appeared concerning morbidity associated with short-term exposure to airborne particles and/or sulfur oxides. Dockery et al. (1982), for example, reported on pulmonary function evaluations carried out for school children in Steubenville, OH as part of the Harvard Six-Cities Study. Pulmonary function was evaluated immediately before and after air pollution episodes in 1978, 1979 and 1980, by relating spirometric measurements (appropriately corrected for height, etc.) to aerometric data (e.g., TSP and SO_2 levels) obtained from state air pollution monitors. Data for each individual child were evaluated. Linear decreases in forced vital capacity (FVC) with increasing TSP concentrations were found, and slopes were determined for linear relationships fitting the data for four different observation periods (fall, 1978; fall, 1979; spring, 1980; fall, 1980). The slope of FVC vs. TSP was calculated for 335 children with three or more observations during any of the four study periods. Of the 335 children examined, 194 were tested during more than one study period. On average, estimated FVC was approximately 2 percent lower following each alert, whereas forced expiratory volume in 0.75 sec ($FEV_{0.75}$) did not change during the 1978 study but was decreased by 4 percent during the 1979 alert. In the spring of 1980, similar declines were seen in FVC and $FEV_{0.75}$ values as were found following the previous alerts, but no significant declines were seen in fall, 1980, when pollutant levels were distinctly lower than for previous alerts (e.g., TSP levels did not exceed 160 $\mu g/m^3$ in fall, 1980). The largest declines in lung function were observed one to two weeks after the episodes. Fifty-nine percent of the children had slopes less than zero (i.e., decreasing FVC with increasing TSP). The median slope for the entire sample was -0.081 mL/μg/m^3, which is significantly less than zero ($p < 0.001$) by a Wilcoxan Signed Rank test. The median FVC vs. SO_2 slope was -0.057 mL/μg/m^3, also significantly ($p < 0.01$) less than zero, but the relationship with mean daily temperature was not significantly less than zero. Similar analyses performed with $FEV_{0.75}$ also showed the relationships (slopes) for SO_2 and TSP to be significantly less than zero.

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*Note: Roth et al. (1986) have recently raised questions regarding how well the health indicator values used in the Lawther morbidity studies reflect actual health status and suggest that associations between temperature and health may be understated in this data set.

Overall, these repeated measurements of lung function showed statistically significant but physiologically small and apparently reversible declines of FVC and $FEV_{0.75}$ levels to be associated with increases of 24-hr mean TSP levels. On days of testing for pulmonary function effects, the TSP levels ranged from 11.0 to 272 μg/m^3 and SO_2 levels ranged from 0.0 to 281 μg/m^3. However, maximum TSP levels of 312 or 422 μg/m^3 occurring in fall, 1978, 2 to 5 days prior to spirometric testing may have contributed to the observed declines in lung function for some children included in data analyses for that period. Similarly, the maximum SO_2 value of 455 μg/m^3 recorded on days immediately preceding the spirometric testing during the Fall, 1979 period may have accounted for observed declines in lung function. The investigators noted that it was not possible to separate the relative contributions of the two pollutants, nor were any thresholds for the observed pulmonary function decrements discernable within the above broad range of TSP and SO_2 levels. Nevertheless, these results appear to demonstrate that small, reversible changes in pulmonary function can occur as the consequence of increased concentrations of TSP and SO_2 somewhere in the above ranges. Whether such pulmonary function changes _per se_ are adverse or can lead to other, irreversible changes or make the lung more susceptible to later insults remains to be resolved. Evaluations of such issues may need to take into account an apparent subset of "responders" within the population of children studied, who showed greater than average declines in lung function in relation to TSP or SO_2 levels. For example, the lowest quartile of slopes of FVC and $FEV_{0.75}$ versus TSP were -0.386 and -0.306 mL/μg/m^3, respectively.

Results consistent with and supportive of the findings of Dockery et al. (1982) have emerged from another recently reported study conducted by Dassen et al. (1986) in the Netherlands. Baseline pulmonary function data were obtained for a sample of more than 600 children during November, 1984. Then, a subset of the same children (N = 62) were retested again in January, 1985, during an air pollution episode when 24-hr mean values for TSP (hi-vol samples), RSP (respirable suspended particulate, $D_{50} \leq 3.5$ by cyclone sampler), and SO_2 (acidimetric technique) measured via a 6-station network all reached the range of 200 to 250 $\mu g/m^3$. Several lung function parameters showed statistically significant average declines of 3 to 5 percent upon second (episode) testing in comparison against each child's own earlier baseline values, including decrements seen on the second day of the episode in both FVC and FEV levels, as well as in measures reflecting small airway functioning (i.e., maximum mid-expiratory flow and maximum flow at 50 percent vital capacity). Declines from their original baseline values for these parameters were still observed 16 days after the episode upon retesting of another subset of the children, but no differences were found between baseline and retest values for a third subset of children reevaluated 25 days after the episode. Given the lack of evident effects at this latter post episode time point, 24-hr mean TSP, RSP, and SO_2 levels measured in the 100 to 150 $\mu g/m^3$ range just prior to the last lung function tests may not be sufficient to cause observable pulmonary function effects in children. Overall, the Dassen et al. (1986) results are very analogous to those found by Dockery et al. (1982) in connection with the Stuebenville episodes. That is, the relative declines in lung function parameters were similar in magnitude (taking into account corrections made for lung growth), and the 2 to 3 week time course for decrements persisting after the episodes were similar.

Mazumdar and Sussman (1983), discussed earlier, not only studied relationships between mortality and measures of PM and SO_x pollution in Pittsburgh, PA during 1972-77, but also included evaluations of morbidity (indexed by emergency hospital admissions) in relationship to daily COH and SO_2 concentrations corrected for temperature and seasonal variations. Significant associations were reported between same-day COH values (which ranged from near 0.0 to 3.5 units) and total morbidity and heart disease morbidity for all ages (1 to 59 yr) and $\geq$ 60 yr age groups, but no consistent statistically significant associations between morbidity categories and same-day SO_2 levels (ranging from near 0 to 0.14 ppm) monitored at the same stations. However, these results cannot be taken as indicative of associations between increased morbidity and elevated PM or SO_2 levels in the Pittsburgh area, given limitations identified earlier in relation to the mortality analyses from the same study, i.e.: (1) inadequate characterization of air pollution concentrations representative of the entirety of Allegheny County from which the morbidity data were drawn, and (2) internal inconsistencies in the results, with various classes of morbidity variously being more strongly associated with SO_2 or COH measured at lower pollution stations than higher pollution stations.

Perry et al. (1983), followed 24 Denver asthmatic subjects from January through March, 1979, using twice daily self-obtained measurements of each subject's peak expiratory flow rates (from Mini-Wright Peak Flow Meters) and recording use of "as-needed" aerosolized bronchodilators and reports of airway obstruction symptoms characteristic of asthma. These measures of morbidity were tested for relationships to air pollutants using a random effects model. Dichotomous, virtual impactor samplers at two fixed monitoring sites provided daily measurements (in $\mu g/m^3$) of inhaled PM (total mass, sulfates, and nitrates), for coarse (2.5 to 15 µm) and fine fractions (<2.5 µm). CO, SO_2, O_3, temperature and barometric pressure were also measured. Of the environmental variables measured, only fine nitrates were significantly associated with increased symptom reports and increased bronchodilator usage. During the course of this study, however, TSP levels were uncharacteristically low. This limits interpretation of the study in relation to PM effects. Use of aerometric data from only two monitoring stations in Denver, with unknown distances in relation to places of residence for subjects matched to the proximal station, also limits the usefulness of the reported findings.

Bates and Sizto (1983, 1986) have also reported results of an ongoing correlational study relating hospital admissions in southern Ontario to air pollution levels. Data for 1974, 1976, 1977, and 1978 were discussed in the 1983 paper. The more recent 1985 analyses evaluated data up to 1982 and showed: (1) no relationship between respiratory admissions and SO_2 or COHs in the winter; (2) a complex relationship between asthma admissions and temperature in the winter; and (3) a consistent relationship between respiratory admissions (both asthma and nonasthma) in summer and sulfates and ozone, but not to summer COH levels. However, Bates and Sizto note that the data analyses are now complicated by long-term trends in respiratory disease admissions unlikely related to air pollution, but they nevertheless hypothesize that observed effects may be due to a mixture of oxidant and reducing pollutants which produce intensely irritating gases or aerosols in the summer but not in the winter. More definitive interpretation of these findings may be limited until additional results are reported from this continuing long-term study.

Goldstein and Weinstein (1986) tested for an association between days with SO_2 peaks above various levels (0.1, 0.3, and 0.5 ppm hourly readings) and days with high numbers of emergency visits for asthmatics at three inner-city municipal hospitals in New York City during 1969 to 1972. Two areas of the city were under study and ambient exposure data were derived from the average of two local air monitoring stations in these areas. No significant associations were found using the two-sided Chi-square test. Potentially confounding factors considered included: day-of-the-week effects, temperature, and trends in asthma following reduction of air pollution in New York City.

Goldstein and Weinstein (1986) stated that the inferences that can be drawn from this ecological study are constrained by certain methodological limitations. For example, they express appropriate concern for the representativeness of the SO_2 exposure data derived from roof top measurements. They also appropriately emphasized that this study does not rule out a relationship between asthma and ambient levels of SO_2 since this ecological approach may be too crude to detect an effect.

Of the newly-reported analyses of short-term PM/SO_x exposure-morbidity relationships discussed above, the Dockery et al. (1982) study provides the best-substantiated and most readily interpretable results. Those results, specifically, point toward decrements in lung function occurring in association with acute, short-term increases in PM and SO_2 air pollution. The small, reversible decrements appear to persist for 2-3 wks after episodic exposures to these pollutants across a wide range, with no clear delineation of threshold yet being evident. In some study periods effects may have been due to 24-hr TSP and SO_2 levels ranging up to 422 and 455 $\mu g/m^3$, respectively. Notably larger decrements in lung function were discernable for a subset of children (responders) than for others. The precise medical significance of the observed decrements per se or any consequent long-term sequalae remain to be determined. The nature and magnitude of lung function decrements found by Dockery et al. (1982), it should be noted, are also consistent with: (1) the recently reported findings of Dassen et al. (1986) for Dutch children; (2) observations of Stebbings and Fogleman (1979) of gradual recovery in lung function of children during seven days following a high PM episode in Pittsburgh, PA (max 1-hr TSP estimated at 700 $\mu g/m^3$); (3) and the report of Saric et al. (1981) of 5 percent average declines in $FEV_{1.0}$ being associated with high SO_2 days (89-235 $\mu g/m^3$).

3.2 EFFECTS ASSOCIATED WITH LONG-TERM EXPOSURES

3.2.1. Mortality Effects of Chronic Exposures

The World Health Organization (1979) notes that, in countries having reliable systems for the collection and analysis of data on deaths, based on cause and area of residence, death rates for respiratory diseases have commonly been found to be higher in urban than in rural areas. Many factors, such as differences in smoking habits, occupation, or social conditions may be involved in these contrasts; however, in a number of countries, a general association between death rates from respiratory diseases and air pollution has been apparent for many decades. Analyses of these data have been of great value as a lead for epidemiologic studies, but the absence of information concerning other relevant variables, such as smoking, and the relatively crude nature of indices of pollution used in many of these studies make them unsuitable for the quantitative assessment of exposure-effect relationships.

The 1982 U.S. EPA criteria document (1982a) noted that certain large-scale "macroepidemiological" studies (or "ecologic" studies as termed by some) have attracted attention on the basis of reported demonstrations of associations between mortality and various indices of air pollution, e.g., PM or SO_x levels. For example, Lave and Seskin (1970) reanalyzed mortality data from England and Wales, and developed multiple regression equations in terms of pollution and socioeconomic indices. Their findings of positive correlations between mortality rates and pollution are of general interest but cannot contribute to the development of dose-response relationships because of inadequate exposure indices used in the analyses. The authors also examined similar data for standard metropolitan statistical areas (SMSAs) in the USA, and in a later paper (Lave and Seskin, 1972) attempted to assess relative effects of air pollution, climate, and home heating on mortality rates. Although equations were obtained relating death rates to measurements of suspended particulate matter and total sulfates (both by high-volume sampler), it is again doubtful whether these can be regarded as valid in the absence of more adequate information on smoking and because of inadequate characterization of exposure parameters.

Other studies reported in further publications (Lave and Seskin, 1977; Chappie and Lave, 1981) extended their earlier analyses. Based on such later work, analogous positive associations between mortality and air pollution variables were reported for the United States. Many criticisms similar to those indicated above for the earlier Lave and Seskin (1970) study apply here. Of crucial importance are basic difficulties associated with all of their analyses in terms of: (1) use of aerometric data without regard to quality assurance considerations, notably including use of sulfate measurements known to be of questionable accuracy due to artifact formation during air sampling; and (2) questions regarding how representative the air pollution data used in the analyses are as estimates of actual exposures of individuals included in their study groups. In some instances, for example, data from a single monitoring station were apparently used to estimate pollution exposures for study populations from surrounding large metropolitan areas.

The 1982 U.S. EPA criteria document (1982a) noted that further difficulties in discerning consistent patterns of association between mortality and air pollution variables are encountered when results of Lave and coworkers are compared with those obtained by others using analogous macroepidemiological approaches. For example, Mendelsohn and Orcutt (1979) carried out regression analyses of associations between 1970 mortality rates (for 404 county groups throughout the United States) and air pollution exposures retrospectively estimated on the basis of 1970 and 1974 annual average pollutant data from air monitoring sites in the same or nearby counties. Their results suggested fairly consistent (though variable) associations between mortality for some age groups (increasingly more positive with age) and sulfate levels but much less consistent and sometimes negative associations with TSP or other pollutants. The combined TSP-SO_4 pollution-health elasticity obtained by Mendelsohn and Orcutt (1979) is similar to that obtained in the earlier studies by Lave and coworkers, all falling in the range of 0.1 to 0.2.

Other results obtained by Thibodeau et al. (1980) in carrying out large scale cross-sectional analyses of the above type indicate that the regression coefficients for mortality relationships with air pollution variables are quite unstable. Also, Lipfert (1980) reported results from an analysis taking into account a smoking index based on state tax receipts, which he interpreted as showing sulfates to be least harmful of seven air pollutants (including SO_2 and TSP), although no adjustments for urban-rural differences in study population residences were used. This is in contrast to unpublished analyses of 1970 United States mortality data by Crocker et al. (1979), which found no significant relationships between air pollution and total mortality when taking into account retrospectively estimated nutritional variables and a smoking index. Also, results of Gerking and Schultze (1981), using the same data base, indicated a significant positive relationship between TSP and total mortality when using an OLS model similar to that of Lave and Seskin (1977) but found negative, though significant, air pollution coefficients after adding smoking, nutrition, exposure-to-cold, and medical-care variables to a two-equation model.

U.S. EPA (1982a) also noted that various criticisms of the above studies have been advanced by authors of the other respective studies, but it was not possible to ascertain which findings may be more valid than others. Thus, although many of the studies qualitatively suggested positive associations between mortality and chronic exposure to certain air pollutants in the United States, many key issues remained unresolved concerning reported associations and whether they are causal or not. Since preparation of the earlier Criteria Document (U.S. EPA, 1982a) additional ecological analyses have been reported regarding efforts to assess relationships between mortality and long-term exposure to particulate matter and other air pollutants.

Chinn et al. (1981), for example, reported an ecological analysis investigating the relationship of mortality to atmospheric smoke and SO_2 in county and London boroughs of England and Wales during 1969 to 1973. Weighted multiple regression analyses showed no significant association between smoke and mortality from respiratory illness. Annual average BS levels were reported to range from 15 to 225 $\mu g/m^3$ and SO_2 levels from 24 to 317 $\mu g/m^3$. The lack of significant association found should not be taken as an indication of no effect at these levels because: (1) the BS readings are derived from the use of mass-reflectance calibration curves with limited or no applicability to the specific geographic locales included in the study; and (2) ecological studies of this type are often very insensitive to small effects of pollution.

Lipfert (1984) conducted a series of cross-sectional multiple regression analyses of 1969 and 1970 mortality rates for up to 112 U.S. SMSA's, using the same basic data set as Lave and Seskin (1977) for 1969 and taking into account various demographic, environmental and lifestyle variables (e.g., socioeconomic status and smoking). Also included in the Lipfert (1984) reanalysis were the following additional independent variables: diet; drinking water variables; use of residential heating fuels; migration; and SMSA growth. New dependent variables included age-specific mortality rates with their accompanying sex-specific age variables. Both linear and several nonlinear (e.g., quadratic or linear splines testing for possible threshold model specifications) were evaluated. Efforts to replicate the basic analyses of Lave and Seskin (1977) and to improve upon the fit of models using various specifications led Lipfert (1984) to conclude that: (1) differences existed between high and low pollution SMSAs unrelated to the magnitude of the air pollution variables, i.e. that there appear to be important variables missing from the specification; (2) correction of errors in the Lave-Seskin data improved the regression fit and significance of some of the coefficients; but (3) it was not possible to conclude whether SO_4 or TSP has a statistically significant effect on total mortality or whether either response is linear.

Lipfert (1984) then introduced additional variables of the type listed above into the reanalysis in hopes of improving the specification and to evaluate possible collinearity with the pollution variables. The fact that some observations were incomplete for some of the newly added variables necessitated the analysis of certain subsets of the original Lave-Seskin data set. Overall, for these reanalyses, in which regressions were extended to include new variables in stepwise fashion (but retaining the 7 Lave-Seskin variables as the first step in each case), adding new variables significantly improved the fit, but several of the original Lave-Seskin variables (including SO_4) became non-significant as the result of the additional variables. Further analyses included regressions for mortality restricted to central city areas versus SMSA-based regressions, with agreement between coefficients for sulfates being quite poor (and negative for central city regressions broken down by age groups <65 or >65 yr). Many of the additional explanatory variables in the above reanalyses (both for central city and SMSA regressions) were found to be statistically significant and were then employed in regressions using total mortality rates adjusted for age, nonwhite population, poverty and cigarette smoking. Results obtained with use of additional explanatory variables and varying model specifications were very mixed: (1) Sulfate coefficients were quite unstable, ranging from near 0.0 to 0.049 (highly significant and corresponding to an elasticity of 6 percent); (2) TSP coefficients were similarly variable, with similar maximum elasticity; (3) In no case were TSP and sulfate variables significant in the same regression; and (4) When the full set of explanatory variables were used with the dummy pollution variables, the coefficients for the pollution variables became more significant. Lipfert (1984), based on these total mortality analyses, concluded that: (1) The Lave-Seskin specification is inadequate and provides misleading results; (2) Using additional explanatory variables improves the fit; (3) The existence of thresholds for the air pollution variables can neither be proved nor disproved; (4) Although difficult to separate SO_4 effects from TSP effects, the TSP coefficients displayed slightly more consistent behavior across all the data sets considered; and (5) Effects for drinking water, ozone, and (to a lesser extent) coal and wood heat warrant further investigation.

Results obtained by Lipfert (1984) with further age- and sex-specific regression analyses for <65 yr old subjects, using all other variables as defined in the above total mortality regressions, produced similar results as for the total mortality analyses. That is, as explanatory variables are added, the pollution variables tend to lose significance and the r^2 values are considerably higher than those of Lave and Seskin (1977), even when using the same specifications. Based on the age- and sex-specific analyses: (1) Sulfate was never significant for males (except for Lave-Seskin specifications) and only occasionally significant for females; and (2) TSP was more often significant for both males and females, especially with threshold specifications. Analogous sex-specific analyses for persons > 65 yr old revealed further interesting results: (1) The migration variable was the single most important variable and the age variable was negative; (2) Sulfate was significant only with the Lave-Seskin specification (both sexes) or with other variables suppressed (females); and (3) TSP was never significant.

In sum, it is quite evident from the above results that the air pollution regression results for the U.S. data sets analyzed by Lipfert (1984) are extremely sensitive to variations in the inclusion/exclusion of specific observations (for central city versus SMSA's or different subsets of locations) or additional explanatory variables beyond those used in the earlier Lave and Seskin (1977) analyses. The results are also highly dependent upon the particular model specifications used, i.e. air pollution coefficients vary in strength of association with total or age-/sex-specific mortality depending upon the form of the specification and the range of explanatory variables included in the analyses. Lipfert's overall conclusion was that the sulfate regression coefficients are not to be taken seriously and, since sulfate and TSP interact with each other in these regressions, caution is warranted for TSP as well.

Ozkaynak and Spengler (1985) have also described recent results from ongoing attempts of a Harvard University group to improve upon some of the previous analyses of mortality and morbidity effects of air pollution in the United States. Ozkaynak and Spengler (1985) present principal findings from a cross-sectional analysis of the 1980 U.S. vital statistics and available air pollution data bases for sulfates, and fine, inhalable and total suspended particles. In these analyses, using multiple regression methods, the association between various particle measures and 1980 total mortality were estimated for 98 and 38 SMSA subsets by incorporating recent information on particle size relationships and a set of socioeconomic variables to control for potential confounding. Issues of model misspecification and spatial autocorrelation of the residuals were also investigated. Results from the various regression analyses indicated the importance of considering particle size, composition, and source information in modeling of PM-related health effects. In particular, particle exposure measures related to the respirable and/or toxic fraction of the aerosols, such as FP (fine particles) and sulfates were the most consistently and significantly associated with the reported (annual) cross-sectional mortality rates. On the other hand, particle mass measures that included coarse particles (e.g., TSP and IP) were often found to be non-significant predictors of total mortality.

The Ozkaynak and Spengler (1985) results noted above for analysis of 1980 U.S. mortality provide an interesting overall contrast to the findings of Lipfert (1984) for 1969-70 U.S. mortality data. In particular, whereas Lipfert found TSP coefficients to be most consistently statistically significant (although varying widely depending upon model specifications, explanatory variables included, etc.), Ozkaynak and Spengler found particle mass measures including coarse particles (TSP, IP) often to be non-significant predictors of total mortality. Also, whereas Lipfert found the sulfate coefficients to be even more unstable than the TSP associations with mortality (and questioned the credibility of the sulfate coefficients), Ozkaynak and Spengler found that particle exposure measures related to the respirable or toxic fraction of the aerosols (e.g., FP or sulfates) to be most consistently and significantly associated with annual cross-sectional mortality rates. It might be tempting to hypothesize that changes in air quality or other factors from the earlier data sets (for 1969-70) analyzed by Lipfert (1984) to the later data (for 1980) analyzed by Ozkaynak and Spengler (1985) and Ozkaynak et al. (1986) may at least partly explain their contrasting results, but there is at present no basis by which to determine if this is the case or which set of findings may or may not most accurately characterize associations between mortality and chronic PM or SO_x exposures in the United States.

Selvin et al. (1984) also used regression analyses applied to ecologic data to study the influence of air quality in the U.S. on mortality. The analyses used 1968-72 mortality data aggregated by county (3082) or by groups of counties comprising 410 1970 Census Public Use Sample (PUS) areas (some of which may be a single heavily populated urban county, e.g. Los Angeles, or several sparsely populated rural counties grouped together). Total mortality, rather than cause-specific, rates were calculated for sex-, race-, and age-specific categories and were then evaluated by regression analyses in relation to air quality values (for TSP, SO_2, and NO_2) extracted from data collected at 6625 monitoring stations during 1974-76. County level aerometric estimates were interpolated from average values at individual monitoring stations, and air pollution estimates for the 410 PUS areas were population-weighted averages of the county level value. Overall, various regression analyses (taking into account numerous control variables) for county-wide or PUS areas in all of the U.S. or broken down into regions (West, South, etc.) yielded extremely mixed results, with both positive and negative coefficients being obtained in various analyses for mortality in relation to TSP, SO_2, and NO_2. The authors: (1) concluded that their results provided no persuasive evidence for links between air quality and general mortality levels; (2) noted that their results were inconsistent with previously published work; and (3) opined that linear regression analyses applied to nationally collected ecologic data cannot be usefully employed to infer causal relationships between air quality and mortality. However, the manner in which the Selvin et al. (1984) study was conducted provides little basis for assigning any credibility to the results obtained, especially in view of: (1) use of 1974-76 air quality data to estimate retrospectively exposures against which to compare 1968-74 mortality data and; (2) use of mortality data aggregated by county or by groups of counties with highly variable relationships between air monitoring locations and the population groups from which the mortality data were drawn.

In addition to ecological or macroepidemiological studies of mortality relationships to chronic air pollution exposures in the U.S., Imai et al. (1986) have recently published analyses of associations between mortality from asthma and chronic bronchitis and air pollution variables in Yokkaichi, Japan. An industrial city on Ise Bay several hundred miles south of Tokyo, Yokkaichi's industrial base and harbor facilities were largely destroyed during World War II. They were later rebuilt to include the establishment in 1957 of a petroleum complex that contained the largest oil-fired power plant in Japan, which burned high-sulfur oil that resulted in large SO_2 emissions and consequent high SO_x concentrations in immediate residential/commercial areas around the harbor. This continued until stringent emission controls were put in place and resulted in dramatic decreases in SO_x concentrations in the highly polluted area around the harbor from 1972 to 1973 and thereafter. Mortality rates for the population in that high pollution area were compared against analogous rates (for bronchial asthma or chronic bronchitis including emphysema, determined from death certificates issued during 1963-83) for people living in less-polluted areas of Yokkaichi. Sulfur oxides levels (measured by the lead peroxide method) averaged across several monitoring sites in the polluted harbor area ranged from around 1.0 to 2.0 mg/day (annual average) during 1964-72 and then steadily declined from somewhat less than 1.0 mg/day in 1973 to less than 0.5 mg/day in 1982. This is in contrast to SO_x levels consistently below 0.3 mg/day (annual average) at 3 monitoring sites in the low pollution areas of the city throughout 1967 to 1982. Annual average levels for other pollutants (NO_2, TSP, oxidants) monitored in the high pollution area were also consistently low, i.e. $\leq$0.02 ppm (NO_2), $\leq$0.05 mg/m^3 (TSP), and $\leq$0.05 ppm (oxidants, daily max hourly values) from 1974 to 1982. Results obtained indicated significant differences between chronic bronchitis mortality for persons $\geq$ 60 yr old in the high pollution area compared against rates for the same age group from the low-pollution control area for 1967-70 and extending into 1971-74, somewhat beyond the point where marked declines became evident in SO_x levels after control measures were implemented. Lagged correlations showed large significant associations between SO_x levels and chronic bronchitis mortality occurring $\geq$1 yr later in the high pollution area (the largest correlations were found for 4-5 yr lags). In contrast, bronchial asthma mortality became relatively higher in the polluted area during the 1967-70 period, and began to decrease thereafter in more immediate response to the improvement in air quality.

These findings, overall, are quite interesting in that they relate mortality changes in populations in circumscribed urban neighborhoods to air pollution indices obtained from monitoring sites spatially located in close proximity to the residences of the population groups for whom mortality rates were determined. Further, consistently elevated mortality for the elderly in the high-pollution area (relative to the control area) was evident across many years while the SO_x concentrations were high, but then declined following reductions in the SO_x levels, thus enhancing the likelihood of a causal relationship between sulfur-containing air pollution and mortality having been detected in the study. However, it is not possible to quantitate with any precision the relative contributions to the observed mortality increases of SO_2 versus sulfates or other sulfur agents (e.g., possibly H_2SO_4 aerosols likely formed in the moist air of the coastal city).

The 1982 EPA document (U.S. EPA, 1982a) also noted that other epidemiological studies have more specifically attempted to relate lung cancer mortality to chronic exposures to sulfur oxides, PM undifferentiated by chemical composition, or specific PM chemical species. However, the 1982 document concluded that little or no clear epidemiological evidence advanced to date substantiates hypothesized links between SO_2 or other sulfur oxides and cancer; nor does there now exist credible epidemiological evidence linking increased cancer rates to elevations in PM as a class, i.e., undifferentiated as to chemical content.

3.2.2. Morbidity Effects of Long-Term Exposures

Increased incidence of respiratory symptoms, disease states or other pulmonary function impairments are likely to be among the effects of long-term exposures to air pollution, since the respiratory system includes tissues that receive the initial impact when toxic materials are inhaled. Numerous studies have been conducted in an effort to relate pulmonary function changes to the presence of PM or sulfur oxides air pollutants in European, Japanese, and American communities. However, few provide more than qualitative evidence relating respiratory symptoms, disease rates or pulmonary function changes to airborne particles and/or sulfur oxides. The few studies evaluated earlier by U.S. EPA (1982a) as providing quantitative evidence for respiratory system effects due to long-term PM and/or SO_x exposure are summarized in Table 2.

One series of studies, reported on from the early 1960s to the mid-1970s, was conducted by Ferris, Anderson, and others (Ferris and Andersen, 1962; Kenline, 1962; Andersen et al., 1964; Ferris et al., 1967, 1971, 1976). The initial study involved comparison of three areas within a pulp-mill town

TABLE 2. SUMMARY OF QUANTITATIVE CONCLUSIONS DRAWN IN U.S. EPA (1982a) FROM EPIDEMIOLOGICAL STUDIES RELATING HEALTH EFFECTS TO CHRONIC EXPOSURE TO AMBIENT AIR LEVELS OF PM AND/OR SO_2***

Type of Study	Effects observed	24-hr average pollutant level (μg/m³) particulate matter BS	 TSP	 SO_2	Reference
Cross-sectional (4 areas)	Likely increased frequency of lower respiratory symptoms and decreased lung function in children in Sheffield, England	230-301*	-	181-275	Lunn et al. (1967)
Longitudinal and cross-sectional	Apparent improvement in lung function of adults in association with decreased PM pollution in Berlin, N.H.	-	180	**	Ferris et al. (1973, 1976)
Longitudinal and cross-sectional	Apparent lack of effects and symptoms, and no apparent decrease in lung function in adults in Berlin, N.H.		80-131	**	Ferris et al. (1973, 1976)

*Note that BS levels stated here in μg/m³ must be viewed as only crude estimates of the approximate PM (BS) mass levels associated with the observed health effects, given ambiguities regarding the use or non-use of site-specific calibrations in Sheffield to derive the reported BS levels in μg/m³.

**Note that sulfation rate methods indicated low atmospheric sulfur levels in Berlin, N.H. during the time of these studies. Crude estimation of SO_2 levels from that data suggest that <25-50 μg/m³ SO_2 levels were generally present in Berlin, N.H., and did not likely contribute to observed health effects.

***Source: U.S. EPA (1982a). Note that the results of new studies published since completion of the earlier 1982 criteria document are described in the text of the present chapter of the Addendum to the U.S. EPA (1982a) document.

(Berlin, New Hampshire). Kenline (1962) reported average 24-h SO_2 levels (estimated from sulfation rates) during a limited summer sampling period (August-September, 1960) to be only 16 ppb and average 24-h TSP levels for the two-month period to be 183 $\mu g/m^3$. In the original prevalence study (Ferris and Anderson, 1962; Anderson et al., 1964), no association was found between questionnaire-determined symptoms and lung function tests assessed in the winter and spring of 1961 in the three areas with differing pollution levels, after standardizing for cigarette smoking. The authors discuss why residence is a limited indicator for exposure (Anderson et al., 1964). The study was later extended to compare Berlin, New Hampshire, with the cleaner city of Chilliwack, British Columbia in Canada (Anderson and Ferris, 1965). Sulfation rates (lead candle method) and dustfall rates were higher in Berlin than in Chilliwack. The prevalence of chronic respiratory disease was greater in Berlin, but the authors concluded that this difference was due to interactions between age and smoking habits within the respective populations.

The Berlin, New Hampshire, population was followed up in 1967 and again in 1973 (Ferris et al., 1971, 1976). During the period between 1961 and 1967, all measured indicators of air pollution fell, e.g., TSP from about 180 $\mu g/m^3$ in 1961 to 131 $\mu g/m^3$ in 1967. In the 1973 follow-up, sulfation rates nearly doubled from the 1967 level (0.469 to 0.901 mg $SO_3/100/cm^2$ day) while TSP values fell from 131 to 80 $\mu g/m^3$. Only limited SO_2 data were available (the mean of a series of 8-h samples for selected weeks). During the 1961 to 1967 period, standardized respiratory symptom rates decreased and there was an indication that lung function also improved. Between 1967 to 1973, age-sex standardized respiratory symptom rates and age-sex-height standardized pulmonary function levels were unchanged. Although some of the testing was done during the spring versus the summer in the different comparison years, Ferris and coworkers attempted to rule out likely seasonal effects by retesting some subjects in both seasons during one year and found no significant differences in test results. Given that the same set of investigators, using the same standardized procedures, conducted the symptom surveys and pulmonary function tests over the entire course of these studies, it is unlikely that the observed health endpoint improvements in the Berlin study population were due to variations in measurement procedures, but rather appear to have been associated with decreases in TSP levels from 180 to 131 $\mu g/m^3$. The relatively small changes observed and limited aerometric data available, however, argue for caution in placing much weight on these findings as quantitative indieffect or no-effect levels for health changes in adults associated with chronic exposures to PM measured as TSP.

The earlier criteria review (U.S. EPA, 1982a) also noted a cross-sectional study conducted by Bouhuys et al. (1978) in two Connecticut towns in which differences in respiratory and pulmonary function were examined in 3056 subjects (adults and children). Hosein, et al. (1977a) reported on aerometric data used in the study, which were obtained at three sites in Ansonia (urban) and four sites in Lebanon (rural) near the residences of study subjects. The TSP levels during the period of the study in Lebanon and Ansonia were 39.5 and 63.1 $\mu g/m^3$ and SO_2 levels were 10.9 and 13.5 $\mu g/m^3$, respectively. Site-to-site variations on the same day were frequently significant in Ansonia and also occurred in Lebanon. During the years 1966-72, annual average TSP levels in Ansonia ranged from 88 to 152 $\mu g/m^3$. No historical data for SO_2 or TSP in Lebanon were provided. Size fractionation (Hosein, et al. 1977b) of a limited number of TSP samples in Ansonia showed 81 percent of the TSP sample to be 9.4 μm or less in diameter. Binder et al. (1976) obtained for 20 subjects in Ansonia one 24-hour measure of personal air pollution exposure for particles ($\leq$7 μm diameter), SO_2, and NO_2. Subjects with smokers in the home were exposed to significantly higher levels than those without such exposure. Personal exposure and outdoor exposures were also significantly different. The mean personal respiratory particle level was 114 $\mu g/m^3$ as compared to the outdoor TSP level of 58.4 $\mu g/m^3$.

An extended version of the MRC Questionnaire was administered via a computer data-acquisition terminal (Mitchell et al., 1976) between October 1972 and January 1973 in Lebanon and from mid-April through July 1973 in Ansonia. For children 7 to 14 yrs, the response rate varied from 91 to 96 percent for boys and girls. For adults (25 to 64 years) the response rate was 56 percent in Ansonia and 80 percent in Lebanon. After analysis of non-responder versus responder differences, the responders were considered to be representative of the total population, although some significant differences were noted between responders and non-responders for some symptom reporting and current smoking in some age groups.

Bouhuys et al. (1978) found no differences between Ansonia and Lebanon for chronic bronchitis prevalence rates but did note that a history of bronchial asthma was highly significant for male residents of Lebanon (the cleaner town) as compared to Ansonia (the higher-pollution area). No differences were observed between the communities for pulmonary function tests adjusted for sex, age, height and smoking habits. However, prevalence for three of five symptoms (cough, phlegm, and plus one dyspnea) were significantly higher for adult non-smokers in Ansonia ($p < 0.001$). The mix of positive and negative health effect results found by this cross-sectional study make it difficult to interpret. Although few air pollution effects, were observed, the statistically significantly increased symptom rates raise questions as to whether some impact on health (due to prior PM exposures, for example) might have occurred. A follow-up longitudinal examination could have determined whether the effects persisted. Also, it may be that the reported effects related more to historical rather than current pollutant levels or to occupational exposures which were not examined.

The 1982 Criteria Document (U.S. EPA, 1982a) further indicated that apparent quantitative relationships between air pollution and lower respiratory tract illness in children were reported by Lunn et al. (1967), who studied respiratory illnesses in 5- and 6-year old school children living in four areas of Sheffield, England. Air pollution levels showed a gradient in 1964 across the four study areas, the mean 24-hour smoke (BS) concentrations ranging from 97 to 301 $\mu g/m^3$. During 1965, annual BS concentrations of smoke were about 20 percent lower and SO_2 about 10 percent higher, but the gradient was preserved for each pollutant. In high-pollution areas, individual 24-h mean BS levels exceeded 500 $\mu g/m^3$ 30 to 45 times in 1964 and 0 to 15 times in 1965 for the lowest and highest pollution areas, respectively. SO_2 exceeded 500 $\mu g/m^3$ 11 to 32 times in 1964 and 0 to 23 times in 1965 for the lowest and highest pollution areas, respectively. Information on respiratory symptoms and illness was obtained by questionnaires completed by parents, by physical examination, and by tests of pulmonary function ($FEV_{0.75}$ and FVC). Socioeconomic factors (SES) were considered in the analyses, but parental smoking and home-heating systems were not. Although some differences in SES between areas were noted, gradients between areas existed even when the groups were divided by social class, number of children in house, and so on. Positive associations were found between air pollution concentrations and both upper and lower respiratory illness. Lower respiratory illness was 33 to 56 percent more frequent in the higher pollution areas than in the low-pollution area ($p < 0.005$). Also, decrements in lung function, measured by spirometry tests, were closely associated with respiratory disease symptom rates.

Lunn et al. (1970) also reported results for 11-year-old children studied in 1963-64 that were similar to those found earlier for the younger group. Upper and lower respiratory illness occurred more frequently in children exposed to annual average 24-h mean smoke (BS) concentrations of 230 to 301 $\mu g/m^3$ and 24-h mean SO_2 levels of 181-275 $\mu g/m^3$ than in children exposed to smoke (BS) at 97 $\mu g/m^3$ and SO_2 at 123 $\mu g/m^3$. This report also provided additional information obtained in 1968 on 68 percent of the children who were 5 and 6 years old in 1963-64. By 1968, the reported BS levels were only about one-half those measured in 1964, SO_2 levels were about 10 to 15 percent below those of 1964, and the pollution gradient no longer existed; so the combined three higher pollution areas were compared with the single original low-pollution area. Lower respiratory illness prevalence measured as "colds going to chest" was 27.9 percent in the low-pollution area and 33.3 percent in the combined high-pollution areas, a difference not statistically significant at $p > 0.05$. Ventilatory function results were similar. Also, the 9-year-old children had less respiratory illness than the 11-year-old group seen previously. Because 11-year-old children generally have less respiratory illness than do 9-year olds, this represented an anomaly that the authors suggested may be due to improved air quality.

These Lunn et al. (1967, 1970) findings have been widely accepted (World Health Organization, 1979; Holland et al., 1979; U.S. EPA, 1982a,b) as valid. On the basis of the results reported, it appears that increased frequency of lower respiratory symptoms and decreased lung function in children may occur with long-term exposures to annual BS levels in the range of 230 to 301 $\mu g/m^3$ and SO_2 levels of 181 to 275 $\mu g/m^3$. However, these are only very approximate observed-effect levels because of uncertainties associated with estimating PM mass based on BS readings. Also, it cannot now be concluded, based on the 1968 follow-up study, that no-effect levels were demonstrated for BS levels in the range of 48 to 169 $\mu g/m^3$ because of: (1) the likely insufficient power of the study to have detected small changes given the size of the population cohorts studied, and (2) the lack of site-specific calibration of the BS mass readings at the time of the later (1968) study. In summary, the Lunn et al. (1967) study provided the clearest evidence cited in the 1982 EPA criteria document (U.S. EPA, 1982a) for associations between both significant pulmonary function decrements and increased respiratory disease illnesses in children and chronic exposure to specific ambient air levels of PM and SO_2.

Since the earlier criteria review (U.S. EPA, 1982a), results of analyses of data from the ongoing Harvard study of outdoor air pollution and respiratory health status of children in six cities in the eastern and midwestern United

States have been reported by Ware et al. (1986). Between 1974 and 1977, approximately 10,100 white preadolescent children were enrolled in the study during three successive annual visits to the cities. On the first visit, each child underwent a spirometric examination and a parent completed a standardized questionnaire regarding the child's health status and other important background information. Most of the children (8,380) were seen for a second evaluation one year later. Measurements of TSP, the sulfate fraction of TSP (TSO_4), and SO_2 concentrations at study-affiliated outdoor stations were combined with data from other public and private monitoring sites to create a record of TSP, TSO_4, and SO_2 levels in each of 9 air pollution regions during a one-year period preceding each evaluation, and for TSP during each child's lifetime up to the time of evaluation.

Analyzing data across all six cities, Ware et al. (1986) found that frequency of chronic cough (see Figure 5) was significantly associated ($p < 0.01$) with the average of 24-hr mean concentrations of all three air pollutants (TSP, TSO_4, SO_2) during the year preceding the health examination. Rates of bronchitis and a composite measure of lower respiratory illness were significantly ($p < 0.05$) associated with annual average particulate concentrations, as well as being related to measures of lifetime TSP concentrations. However, within the individual cities, temporal and spatial variation in air pollutant levels and symptom or illness rates were not significantly associated. The history of early childhood respiratory illness for lifetime residents was significantly associated with average TSP levels during the first two postnatal years within cities, but not between cities. Furthermore, pulmonary function parameters (FVC and FEV_1) were not associated with pollutant concentrations during the year immediately preceding the spirometry test (see Figure 6) or, for lifetime residents, with lifetime average concentrations, although Ferris et al. (1986) reported a small effect on lower airway function (MMEF) related to FP concentrations.

Overall, these results appear to suggest that risk may be increased for bronchitis and some other respiratory disorders in preadolescent children at moderately elevated TSP, TSO_4 and SO_2 concentrations, which do not appear to be consistently associated with pulmonary function decrements. However, the lack of consistent significant associations between morbidity endpoints and air pollution variables within individual cities argues for caution in interpreting the present results. For example, it might be argued that the non-significant

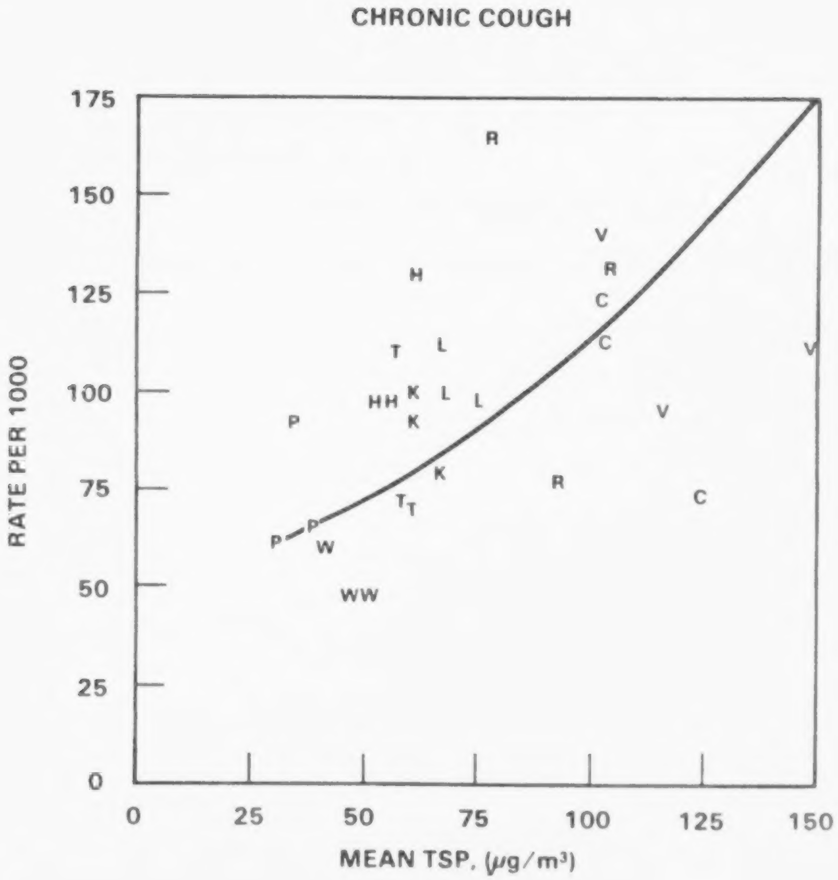

Figure 5. Adjusted frequency of cough for the 27 region-cohorts from the Six-Cities Study at the second examination plotted against mean TSP concentration during the previous year, with between-cities regression equation. LEGEND: P=Portage, T=Topeka, W=Watertown, C=Carondolet, L=Other St. Louis, R=Steubenville Ridge, V=Steubenville Valley, K=Kinston, H=Harriman.

Source: Ware et al. (1986).

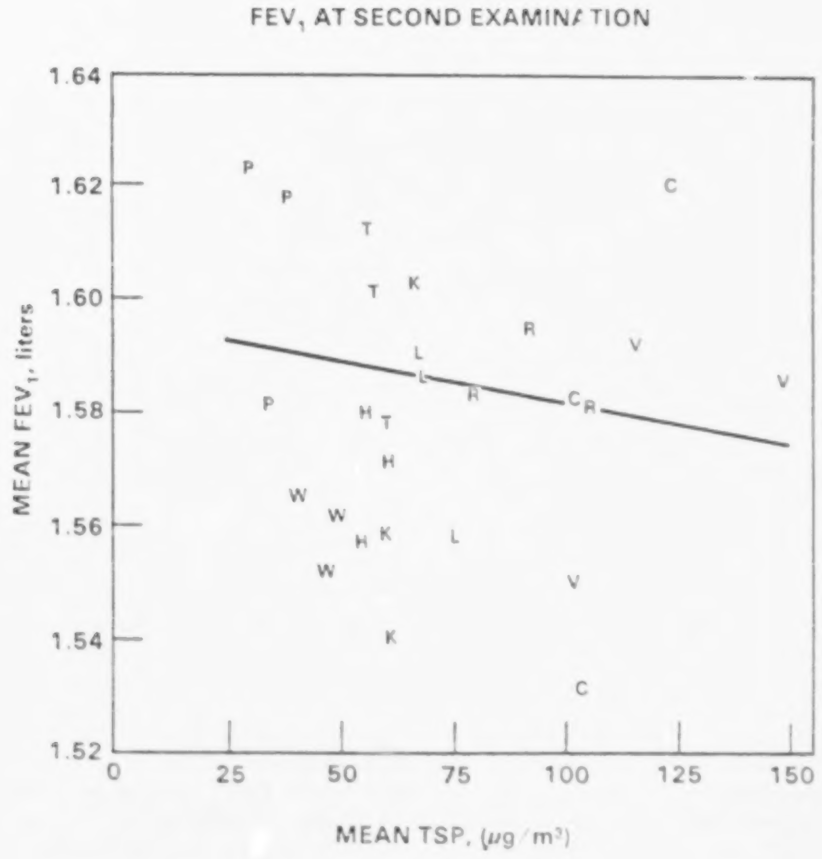

Figure 6. Adjusted mean percent of predicted FEV$_1$ at the first examination for the 27 region-cohorts from the Six-Cities Study plotted against mean TSP concentration during the previous year, with between-cities regression equation. The slope is not significantly different from 0. LEGEND: See Figure 5.

Source: Ware et al. (1986).

associations within cities but significant symptom increases in relation to air pollutant gradients across the cities may reflect spurious correlations across the cities. On the other hand, the within city variation in air pollutant gradients and/or size of study populations within particular cities may not be sufficiently large to detect associations between the health endpoints and air pollutant variables included in the analyses. Also, the PM indices employed in the analyses (e.g., TSP, etc.) may provide a "diluted" measure of exposure to the most highly toxic PM components (e.g., FP or small coarse-mode particles). In fact, the reported stronger associations between TSO_4 levels and other measures of ambient air FP concentrations are highly suggestive of possible associations between health effects observed in the Ware et al. (1986) study and exposure to small particles in contemporary U.S. atmospheres. Available data (Spengler and Thurston, 1983) from air monitors sampling inhalable particulates (IP; <15 µm) in the same cities included in the Harvard Six Cities Study analyses discussed here indicate IP mass annually averaged from approximately 20 to 60 $\mu g/m^3$. This suggests that the observed health effects noted above may be associated with annual average IP (<15 µm) concentrations below 60 $\mu g/m^3$. However, full interpretation of the strength and significance of these findings is difficult at this point, in light of further follow-up of these children still being in progress and the expectation that longitudinal analyses will later be carried out which will relate health data to more extensive aerometric data (including such data collected in later years).

In another series of studies conducted during the last few years, Ostro and co-workers evaluated relationships between air pollution indices for 84 standard metropolitan statistical areas (SMSA's) mostly of 100,000 to 600,000 people in size, and indices of acute morbidity effects, using data derived from the National Center for Health Statistics (NCHS) Health Interview Survey (HIS) of 50,000 households comprising about 120,000 people (Ostro, 1983; Hausman et al., 1984; Ostro, 1987).

Ostro (1983) used HIS data to assess the prevalence of illness and illness-related restrictions in activity in the United States. Data on either restricted activity days (RADs) or work loss days (WLDs) were aggregated over a year, and correlated with annual TSP levels, controlling for temperature, wind, precipitation, population density, smoking, etc. Using the 1976 survey, a significant relationship between TSP and both outcomes was found, with RAD's showing a more significant relationship. The explained variation was much higher for RADs than for WLDs. This is expected since the decision to take a day off from work depends on many idiosyncratic factors besides if illness is present. The average of air pollution monitors for each city was used, rather than aerometric data aggregated for smaller geographic units in relationship to individuals residing nearby for whom HIS data were included in the analysis. The use of city wide TSP data therefore increased possible error in the exposure variable, but this would more likely bias the results toward zero, rather than towards finding a significant effect. The Ostro (1983) analysis was also only for one year of data, and thus was unable to demonstrate consistency across years. On the other hand, the use of 84 cities in the Ostro (1983) analysis reduced the chance that the particular choice of cities spuriously induced a relationship between air pollution and morbidity due to some omitted cofactor. In sum, this first paper suggested a potential relationship between morbidity and air pollution, which must be viewed with caution because of the ecological nature of the data, the less than perfect fit of the annual pollution and acute morbidity variables, and because of the possibility that results in one year could have occurred by chance.

The Hausman et al. (1984) paper analyzed the same data, but made three important methodological advances. It used a Poisson specification for the model, used a fixed effects model that only looks at deviations from the city mean levels of illness, and used short-term pollution as the exposure variable. Poisson analysis is appropriate for analyzing low probability events, which is the case with these morbidity symptoms. The fixed effect model effectively controls for differences between cities in morbidity levels. This avoids the potential bias of attributing intercity differences in disease rates to intercity differences in pollution. Two-week average TSP levels are used as the exposure variable. Significant associations between pollution levels and RADs or WLDs were still found. The magnitude of the within city effects was similar to the magnitude of the between city effects seen earlier. Again demographic factors were controlled for on an individual basis, along with climatic conditions. This analysis considerably strengthens the plausibility of the association, particularly because of the within city effect. However, it still only analyzed data for one year, which may be anomalous.

In the most recent analyses reported, Ostro (1987) applied the Hausman et al. (1984) techniques to analyze HIS results from 1976 to 1981 in relation to estimates of fine particle (FP) mass. That is, for adults aged 18 to 65, days of work loss (WLDs), restricted activity days (RADs) and respiratory-related restricted activity days (RRADs) measured for a two-week period before the day of the survey were used as measures of morbidity and analyzed in relation to estimated concurrent two-week averages of FP or lagged in relation to estimated 2-wk FP averages from 2 to 4 weeks earlier. The FP estimates were produced from the empirically derived regression equations of Trijonis. These equations, as used here, incorporated screened airport data and two-week average TSP readings at population-oriented monitors, using these data taken from the metropolitan area of residence. Various potentially confounding factors (such as age, race, education, income, existence of a chronic health condition, and average two-week minimum temperature) were controlled for in the analyses. Various morbidity measures (WLDs, RADs, RRADs), for workers only or for all adults in general, were consistently found to be statistically significantly ($p < 0.01$ or < 0.05) related to lagged FP estimates (for air quality 2 to 4 weeks prior to the health interview data period), when analyzed for each of the individual years from 1976 to 1981. However, less consistent associations were found between the health endpoints and more concurrent FP estimates.

The approach employed by Ostro to estimate PM levels introduces into his analyses a number of uncertainties, such as those inherent in airport visibility measurements, FP/visibility relationships, and TSP monitoring limitations (most notably, use of the Trijonis equations characterizing FP relationships to visibility in northeastern U.S. areas may not be appropriate for western U.S. cities); and use of single average pollutant levels to estimate exposures for an entire city's population. Use of the spatially averaged indicator over time within a specific area should reduce some of these uncertainties, but it is unlikely that more than qualitative relationships between PM levels estimated in this fashion and morbidity effects could be derived. Additional uncertainties derive from use of the HIS data base, with the vast majority of data points being "0", representing no incidences of indicator effects being recalled in the prior two weeks. However, use of the Hausmann et al. (1984) statistical approach should have adequately dealt with this problem.

The overall patterns of results obtained from the reported analyses are interesting but difficult to interpret. They may suggest that acute morbidity effects are associated with fine-mode particle exposures occurring 2-4 weeks earlier, but less so with immediately prior FP exposures. Such a possibility cannot be ruled out in view of findings reported by other investigators regarding lag structures in data bases relating mortality or morbidity to PM exposures. Nevertheless, these Ostro analyses have found consistent associations between PM and morbidity measures for adults that are reasonably consistent between and within contemporary American cities. As such, the results tend to reinforce the plausibility of the Ware et al. (1986) findings of associations between morbidity measures in children and PM concentrations found in contemporaneous American urban air sheds. However, the Ostro analyses do not allow for the estimation of quantitative relationships between morbidity effects and more usual 24-hr or annual average direct gravimetric measures of particulate matter air pollution (e.g., TSP, PM_{10}, etc.).

In another new American study, by Schenker et al. (1983), respiratory symptom questionnaires were administered to 5557 adult women in a rural area of western Pennsylvania. Air pollution data (including SO_2 but not PM measurements) were derived from 17 air monitoring sites and stratified in an effort to define low, medium and high pollution areas. The means of 4-yr (1975-1978) annual average SO_2 levels in each stratum were 62, 66, and 99 $\mu g/m^3$, respectively. Risks for respiratory symptoms were assessed by a multiple logistic model that controlled for several potentially confounding factors (e.g., smoking) and used estimated air pollution concentrations at population-weighted centroids of 36 study districts (i.e., the concentrations were derived from another model which weighted observed monitoring data for distance from the district centroid and corrected for terrain effects). The risk of "wheeze most days or night" in nonsmokers residing in the high- and medium-pollution areas was 1.58 and 1.26 ($p = 0.02$), respectively, in relation to the low-pollution area. For residents living in the same location for ≥ 5 yr, these relative risks were 1.95 and 1.40 ($p < 0.01$), and increased risk of grade 3 dyspnea in nonsmokers was associated with SO_2 levels at $p \leq 0.11$. However, no significant association was observed between cough or phlegm and air pollution variables. The results of this study, while suggesting that wheezing may be qualitatively associated with ambient exposure to SO_2, are difficult to accept in light of: (1) the very limited gradient of annual-average SO_2 levels across which health effects were reported to have been detected (associations with higher level exposures versus distinctly lower SO_2 concentrations would be more credible); (2) the very rough estimation of SO_2 exposure concentrations by means of model calculations; and (3) the lack of evaluation of possible PM or short-term SO_2 peak contributions to the evaluated health effects.

Several other recent studies have been reported that evaluated PM and/or SO_x effects in populations residing in the southwestern United States. In one, Chapman et al. (1985) conducted a survey in early 1976 regarding the prevalence of persistent cough and phlegm (PCP) among 5,623 young adults in four Utah communities stratified to represent a gradient of sulfur oxides exposures. Community-specific mean SO_2 levels had been 11, 18, 36 and 115 $\mu g/m^3$ during the 5 years prior to the survey and corresponding mean sulfate levels were 5, 7, 8, and 14 $\mu g/m^3$. No gradients of TSP or suspended nitrates were observed across the communities. Aerometric data were obtained from monitors sited at ground level. Differences along the sulfur oxides gradient were tested by chi-square statistics, and data were also analyzed by constructing categorical logistic regression models that treated PCP as the dependent variable and controlled for numerous potentially important factors (e.g. smoking, age, SES, etc.). For nonsmoking mothers, PCP prevalence was 4.2 percent in the high-exposure community and ~2.0 percent in all other communities. For non-smoking fathers, the PCP prevalence was 8.0 percent in the high pollution community and 3.0 percent elsewhere, while the PCP prevalence was less strongly associated with ambient sulfur oxides exposures for smoking fathers. Overall, intercommunity prevalence differences were significant at $p < 0.05$ for all the above groups except smoking fathers. The categorical logistic regression model yielded similar results, providing evidence suggestive of increased cough and phlegm being associated with annual average 115 $\mu g/m^3$ SO_2 levels and/or 14 $\mu g/m^3$ sulfate levels. There is much to argue for acceptance of the reported results from this study, including use of aerometric data from monitors situated in close proximity to study subjects' homes and nearly equivalent response rates on the health questionnaire across the communities sampled.

Dodge (1983) studied the respiratory health and lung function of Anglo-American children (grades 3 to 5) residing in an Arizona smelter community versus such children residing in another small Arizona community free of smelter air pollution. Cough prevalence was 25.6 percent in the smelter town children and 14.3 percent in the non-smelter groups ($p < 0.05$). Baseline pulmonary function at the outset of the study was equal in the two groups, and over the four years of the study, lung function growth (measured in terms of FEV_1 after 4 yr. of study minus predicted FEV_1) was also equal between the two groups. During the study, annual average SO_2 levels were 55 and 48 $\mu g/m^3$ at company and state monitoring sites, respectively (highest 24-hr SO_2 levels were 611 and 524 $\mu g/m^3$, respectively, at the company and state sites). Annual average TSP was 28 $\mu g/m^3$ in the smelter community. These results suggest that smelter community children had more cough than the control group children but no evident differences in lung function. However, it is difficult to ascribe the reported effects specifically to SO_2 or TSP (although the very low levels of the latter are unlikely to account for the effects).

Dodge et al. (1985) more recently reported on a longitudinal study of children exposed to markedly different concentrations of SO_2 and moderately different levels of particulate sulfate ($SO_4^=$) in Southwestern U.S. towns. Four groups of subjects lived in two areas of one smelter town and in two other towns, one of which was also a smelter town. In the highest pollution area, the children were exposed intermittently to high SO_2 levels (peak 3-hr $\bar{x}$ exceeded 2,500 $\mu g/m^3$ or ~1.0 ppm) and moderate particulate SO_4 levels ($\bar{x} = 10.1$ $\mu g/m^3$). When children were grouped by the four observed pollution gradients, the prevalence of cough (measured by questionnaire) correlated significantly with pollution levels (trend chi-square = 5.6; $p = 0.02$). No significant differences occurred among the groups of subjects over 3 years, and pulmonary function and lung growth over the study were roughly equal over all groups. The results tend to suggest that intermittent high level exposures to SO_2, in the presence of moderate particulate sulfate levels, produced evidence of bronchial irritation (increased cough) but no chronic effect on lung function or lung function growth. It is difficult to quantitate the SO_2 levels specifically associated with the observed effects, although the intermittent high level exposures to ~1.0 ppm (3 hr averages) mentioned earlier are likely implicated. Note that SO_2 levels for the higher polluted smelter town annually averaged 103 ± 282 (S.D.) $\mu g/m^3$ (indicating wide variability in the one hr mean levels) versus 14 $\mu g/m^3$ in the lesser polluted town. Other measured air pollutants, e.g. TSP, differed little between the high and low pollution areas (24-hr TSP $\bar{x} = 52$ and 58 $\mu g/m^3$, respectively). The observation of increased cough but lack of lung function changes in children comports well with the findings of Ware et al. (1986).

Lebowitz et al. (1982) studied 117 families in Tucson, Arizona, selected from a stratified sample of families in geographical clusters from a representative community population included in an ongoing epidemiologic study. Both asthmatic and non-asthmatic families were evaluated over a two year period, using daily diaries; and the health data obtained were related to various indices of environmental factors derived from simultaneous micro-indoor and outdoor monitoring in a representative sample of houses for air pollutants, pollen, fungi, algae and climate. Macromonitoring of air pollutants and pollen was carried out simultaneously. The data were mainly evaluated in terms of statistical techniques employing contingency tables and frequency distributions using SPSS programs. Two-month averages of indoor TSP ranged from 2.1 to 169.6 $\mu g/m^3$. Cyclone measurements of respirable particulate (RSP) ranged from below readable limits up to 28.8 $\mu g/m^3$. CO and NO_x measurements were also taken, but no SO_2 monitoring was reported. Suspended particulate matter and pollen were reported to be related to symptoms in both asthmatics and non-asthmatics, but the authors reported that the statistical analyses used were all qualitative (becase of low sample size) and statistical significance was not computed.

In a recently published Canadian study, Pengelly et al. (1986) reported results for an ongoing study of associations between particle size and respiratory health in children of Hamilton, Ontario. From 1979 to 1982, a cohort of approximately 3500 elementary school children was studied by determining each child's health history and respiratory symptoms by means of a questionnaire administered to their parents. Also, pulmonary function tests were conducted on the children at school. Particle size and concentrations were determined by using two networks distributed across the city, one consisting of 7 to 9 Anderson 2000 Cascade impactors and another of 27 hi-vol TSP samplers. Smoking, use of gas for cooking, SES and other potentially confounding factors were assessed by parental questionnaire and controlled for in statistical analyses, i.e., stepwise multiple regression techniques (linear for continuous dependent variables and logistic for binary dependent variables).

In the present report, Pengelly et al. (1986) focused on two indicators of respiratory health (cough and bronchitis episodes) and two indicators of pulmonary function (peak expiratory flow or PF and MEF_{75}), both adjusted for body size. Logistic regression analyses found no significant associations between cough or bronchitis episodes and air pollution indices, correcting for other factors. Both peak flow and MEF_{75} (adjusted for height) were reported to be significantly associated with the presence of fine particles. However, the fine fraction (FF) was estimated by adding results for samples collected by the lower stages of a cascade impactor (nominally reflecting sizes <3.3 μm). Based on particle bounce problems associated with this impactor (see discussion in Chapter 1) and comparison measurements made by the authors in Hamilton between dichotomous fine (<2.5 μm) and the cascade FF, additional coarse material >3.3 μm was probably also included in the FF measured by Pengelly et al. (1986). Overall the FF mass was more than double the dichotomous sampler fine mass.

Also since preparation of the earlier criteria review (U.S. EPA, 1982a), additional analyses of health effects relationships to PM and SO_x air pollution in European cities have emerged. Some of the new European work includes longitudinal analyses reported by van der Lende et al. (1986) as being conducted in regard to evaluating relationships between prevalence of respiratory symptoms and pulmonary function decline and variations in air pollution in two areas of The Netherlands. That is, health measurements were obtained from cohorts of approximately 2000 men and women (aged 15 to 64 years), residing in a highly polluted area (Vlaardingen) or a non-polluted rural area (Vlagtwedde), with subjects being followed and examined at intervals of three years. Over the course of the study, air pollution levels (PM measured as British smoke, SO_2, etc.) remained consistently very low in the latter area, whereas pollution levels declined over time in the former, highly polluted area. Van der Lende et al. (1986) noted that in a previous publication, they reported both a significantly higher prevalence of respiratory symptoms in the polluted area and also a greater decline there in pulmonary function (based on four consecutive studies over a 9-year period). In the present update paper (van der Lende et al., 1986), further findings are provided regarding associations between respiratory symptoms and pulmonary function decline and air pollution after six consecutive studies covering a 15-year period. The results, termed "preliminary" by the authors, provide some indications of more respiratory symptoms and greater pulmonary function declines in the polluted area than the control, non-polluted area. However, as currently available, the reported results do not allow for any quantitative conclusions to be clearly drawn regarding PM levels associated with observed health effects.

In another study (PAARC, 1982a,b; Lellouch, 1986) relationships between atmospheric pollution and chronic or recurrent respiratory diseases were evaluated from 1974 to 1976 as part of a French national survey in 28 areas of 7 cities and a newly industrialized region. The following pollutants were measured: SO_2 (specific-SP and acidimetric-AF methods); suspended particles (smoke and modified OECD gravimetric methods); nitrogen oxides (NO and NO_2

measured by modified Griess-Saltzmann method); and sulfates (measured by colorimetry after reduction). Samples were obtained over 24 hr. periods, but for the gravimetric measures (48 to 96 h), from 1974-76 except for one summer month each year and except for the sulfates which were determined only during the last half of the study and only in one part of the study zones. Twenty-eight study zones were defined to include 2-4 groups of ~1000 people in different cities exposed to pollution that differed as much as possible in quality and quantity (estimated from earlier aerometric data from 1971-72). Zones included populations situated within 0.5 to 2.3 km ($\bar{x}$ = 1.3 km) of air monitoring stations located 2-4 m above ground level in the center of each zone. National meteorological services supplied climatic data (e.g., temperature and humidity) taken at a station best characterizing each city (usually an airport, sometimes far from the zones investigated), and laboratory analyses for the air pollutants measured were carried out by laboratories in each city studied but for sulfates done at a single laboratory. Means for daily data for the pollutants studied were calculated for 1974-76 (where values came from data accumulated over several days, it was assumed the pollution was the same on each day). The extreme mean daily concentrations from various zones were: 13 and 127 $\mu g/m^3$ for SO_2 (AF), 22 and 85 $\mu g/m^3$ SO_2 (Sp); 18 and 152 $\mu g/m^3$ (smoke); 45 and 243 $\mu g/m^3$ (gravimetric), 7 and 145 $\mu g/m^3$ (NO); and 12 to 61 $\mu g/m^3$ (NO_2).

As for health evaluations, ventilatory function was measured in both men and women aged 25 to 59 and children aged 6 to 10 and respiratory symptoms were ascertained by standardized questionnaire. The results presented by PAARC (1982a,b) were for ~20,300 subjects from 20 zones (response rates varied from 70 to >90 percent in the included zones). Analyses of covariance were used for FEV results and logistical regression for the analysis of symptoms scores, taking into account control factors such as smoking and socioeconomic status. It should be noted that efforts were made to standardize the health endpoint measurements by common training of personnel carrying out testing in various zones and use of standard protocols.

The results of the study were reported by PAARC (1982b) as follows: (1) Among both male and female adults, SO_2 concentrations are significantly associated with the prevalence of lower respiratory disease (LRD) symptoms; (2) Among children, SO_2 is associated with the prevalence of upper respiratory disease (URD) symptoms; (3) For both adults and children, $FEV_{1.0}$ varied negatively in relation to elevations in SO_2 levels; and (4) No other pollutants were associated with ventilatory function or the prevalence of respiratory symptoms. More specifically, SO_2 concentrations were significantly correlated ($r > 0.44$) with incidence of cough, expectoration, and LRD symptoms in men and with LRD incidence in women ($r = 0.49$); and SO_2 correlated ($r = 0.53$) significantly with URD in children. It was noted that, whereas the above results emerged from analyses including data drawn from across cities, the gradient of SO_2 effects on symptom rates was not always evident within the same city (an analogous situation to findings reported by Ware et al., 1986, based on data from six American cities). Similarly, the gradients emerging from regressions across cities for relationships between SO_2 and $FEV_{1.0}$ measures for men ($r = -0.52$), women ($r = -0.67$) and children ($r = -0.70$) were not always evident from data within all individual cities. In contrast to the SO_2 results, very mixed correlations (some positive and some negative, but none significant) were found between symptoms and measures of PM (smoke or gravimetric) and nitrogen oxides (NO, NO_2). Also, PAARC (1982b) reported that the correlations between $FEV_{1.0}$ and PM or nitrogen oxides measures were positive (some significantly so for NO or NO_2); i.e., they implied improved lung function as airborne particle or nitrogen oxides levels increased. The Lellouch (1986) publication, apparently based on final data analyses utilizing approximately the same number of subjects as noted in the PAARC (1982b) report, emphasized the findings noted above for SO_2-related health effects but stated that no clear correlations were observed for any other pollutants (i.e. sulfates, particulate matter, or nitrogen oxides).

The results from the PAARC study (PAARC, 1982a,b; Lellouch, 1986) are interesting but challenging in terms of interpretation. The study appears to have ensured that aerometric data from the sampling stations used would be reasonably well representative of the surrounding study populations in the various zones, a definite strong point of the study. Similarly, efforts to standardize measurements of health endpoints across the different cities is another strong point. Also, in the case of the SO_2 measurements, acceptable analytical techniques were used and periodic intercomparisons made between laboratories, thus enhancing the credibility of the SO_2 aerometric data. Much less confidence can be placed in the data derived for particulate matter, however, in view of the use of smoke readings and/or gravimetric readings that varied for 48 to 96 h periods as the basis for generating estimated particle concentrations to compare across cities. It is doubtful that any adequate comparison could be made, then, across cities in terms of relationships between either symptoms or pulmonary function measures and PM estimates; analyses relating such health endpoints to PM measures within individual cities (not reported in PAARC, 1982a,b) might be more credible, but this remains to be evaluated. As for the significant associations between SO_2 and health endpoints reported by PAARC (1982a,b), several factors limit full acceptance of the reported findings, eg: (1) the SO_2 and PM indices were only tested in separate regression analyses; (2) the associations for SO_2 and lung function changes were significant for only one of the two types of SO_2 measurement methods used; and (3) other uncertainties are introduced by the lack of control for seasonal effects and parental smoking in the analyses of childrens' data.

In another European study from the Commission of the European Communities (Florey et al., 1983) reported since the 1982 U.S. EPA criteria document was prepared, various health endpoints in children (6-11 yrs old) were evaluated in relation to air pollution in 19 geographic areas located in several different European Community countries. Data were obtained on 22,337 children and included information on respiratory symptoms obtained by questionnaire and pulmonary function measurements (peak expiratory flow rate measured by Wright peak flow meters). Efforts were made to standardize health measurements and protocols across all study areas. SO_2 concentrations were determined (using six different analytical methods) and particulate pollution was measured by smoke methods in some countries and by unspecified gravimetric methods in a few other ones. Side by side monitors were set up at 20 sites to help provide a basis for calibration across sites; these 20 "comparison" monitoring stations standardly used the British smoke method for PM and acidimetric method for SO_2. Significant associations emerged from analyses within some individual countries, but differed greatly from one country to another. In three countries, a composition variable called chronic non-specific lung disease (CNSLD) was highly significantly correlated positively with smoke, but the magnitude of the effects differed by a factor of about seven. The range of annual smoke levels was about the same in all three countries, about 15-40 $\mu g/m^3$. In four countries, there were significant associations with SO_2, but two of these were negative. In those with positive correlations annual median SO_2 levels were 60-160 $\mu g/m^3$, and for those with negative associations they were 20-120 $\mu g/m^3$, making it likely that the SO_2 results reflected chance variations rather than actual pollution effects. However, no significant relationships between health effects and particulate pollution were found when data from across countries were pooled. The reported results are difficult to interpret. The Commission of the European Communities (Florey et al., 1983) report noted that annual average levels of smoke greater than 140 $\mu g/m^3$ in the presence of SO_2 at $\geq$180 $\mu g/m^3$ have been found by other studies to be levels above which consistent positive associations between health effects and air pollution are detectable. These levels are higher than any measured in the present study, and this might explain the lack of consistent effects observed from city to city or when data were analyzed across all cities. The results of analyses for data within a given city may warrant further, more detailed evaluation and may yield useful information on quantitative exposure-effect relationships. However, given the great difficulty noted by the Commission of the European Communities (Florey et al., 1983) report in deriving bases for comparing air quality measurements for PM and SO_2 across different cities it is dubious that useful quantitative conclusions can be drawn from analyses of data combined across cities. This is especially the case in view of only limited calibration of smoke readings against gravimetric measurements by collocated gravimetric devices in the various countries.

Muehling et al. (1985) also studied the relationship between croup and obstructive bronchitis of German children taken to clinic versus the level of air pollutants of their residential areas. They show in this retrospective study that the incidence of these two diseases was greater in the area with higher SO_2 and dustfall levels. Several important confounding factors were examined (i.e., infection incidence, meteorological parameters, social status, and distance from clinic). Quarterly average values of SO_2 and dustfall were provided by the county of Nord Rhein in Westphalia. The authors state that their results clearly show that the disease frequency depended on whether the children lived in an area of high or low SO_2 and dustfall levels, but noted that it cannot be clearly stated whether or not the measured emissions are the actual cause of any increased morbidity.

Wojtyniak et al. (1984) studied the symptoms of persistent cough and phlegm, bronchitis, and reduced ventilatory capacity in Cracow, Poland. This cross-sectional study used questions based on the MRC questionnaire. An extensive monitoring network of 20 sampling stations covered the entire area of the city. The city was divided into two parts: the city center with suspended particulate levels averaging 180 μg/m^3 and SO_2 levels of 114 μg/m^3, and the remaining areas having suspended particulate levels averaging 109 μg/m^3 and SO_2 levels averaging 53 μg/m^3. Multiple logistic regression models were used to test for the effects of air pollution, age, smoking history, and other factors. As expected, smoking history was a highly significant determinant, but high exposure to air pollution did result in 2.3 times ($0.05 < p < 0.10$) the risk of exacerbated symptoms in men. In women, the prevalence of exacerbated symptoms was related to indoor air pollution resulting from coal combustion in stoves. Because only two pollution exposure areas were used, it was impossible to separate the effects of particulate matter and sulfur dioxide. The study may also minimize the effect of pollution because of confounding of smoking and because of the lack of a true "clean" control area.

In summary, of the numerous new studies published on morbidity effects associated with long-term exposures to PM or SO_x, only a few provide potentially useful results by which to derive quantitative conclusions concerning exposure-effect relationships for the subject pollutants. The Ware et al. (1986) study, for example, provides evidence of respiratory symptoms in children being associated with particulate matter exposures in contemporary U.S. cities without evident threshold across a range of TSP levels from ~30 to 150 μg/m^3, with more marked effects notable in the 60-150 μg/m^3 range in comparison to lower levels. The increase in symptoms appear to occur without concomitant decrements in lung function among the same children. The medical significance of the observed increases in symptoms unaccompanied by decrements in lung function remains to be fully evaluated but is of likely health concern. Caution is warranted, however, in using these findings for risk assessment purposes in view of the lack of significant associations for the same variables when assessed from data within individual cities included in the Ware et al. (1986) study. The findings derived from the series of studies by Ostro (Ostro, 1983; Hausman et al., 1984; Ostro, 1987), qualitatively indicative of morbidity effects in adults being associated with PM exposures with U.S. cities, tend to support the plausibility that the associations observed by Ware et al, (1986) reflect actual morbidity effects in children due to contemporaneous PM and/or SO_2 exposures in U.S. cities.

Other new American studies provide evidence for: (1) increased respiratory symptoms among young adults in association with annual-average SO_2 levels of ~115 μg/m^3 (Chapman et al., 1985); and (2) increased prevalence of cough in children (but not lung function changes) being associated with intermittent exposures to mean peak 3-hr SO_2 levels of ~1.0 ppm or annual average levels of ~103 $\mu g/m^3$ (Dodge et al., 1985). It is difficult in regard to each of these two studies, however, to determine if the reported effects are due to repeated high-level intermittent exposures to SO_2 or to more chronic low level exposures to SO_2 or its transformation products.

Results from one European study (PAARC, 1982a,b) also tend to suggest that increased lower respiratory disease symptoms and decrements in lung function in adults (both male and female) may be associated with annual average SO_2 levels in the range from about 25 to 130 $\mu g/m^3$. In addition that study suggests that upper respiratory disease and lung function decrements in children may also be associated with annual-average SO_2 levels across the above range. The SO_2-morbidity effects associations reported by PAARC (1982a,b), however, cannot be fully accepted in view of several factors discussed earlier, e.g. internal inconsistencies between results obtained with analyses using different SO_2 measurement data and lack of control for some important potentially confounding factors in certain of the analyses yielding significant results.

3.3 SUMMARY AND CONCLUSIONS

As indicated earlier, although key conclusions from the 1982 criteria document (U.S. EPA, 1982a) are concisely summarized at the outset of various chapter subsections, the main focus of this chapter is on the evaluation of epidemiological information on the health effects of PM and SO_x newly available since preparation of the 1982 document. Furthermore, major emphasis has been placed in this chapter on identification of the newer epidemiological studies or analyses which provide quantitative information pertinent to delineation of exposure-effect or exposure-response relationships.

Table 3 summarizes key conclusions drawn from those newer studies or analyses evaluated in the present chapter as providing the most pertinent and useful quantitative evidence for mortality or morbidity effects associated with short-term human exposures to PM or SO_2.

Taking into account the first category of studies in Table 3 and various considerations discussed above in this chapter, the following conclusions appear to be warranted based on the earlier criteria review (U.S. EPA, 1982a) and the present evaluation of newly available analyses of the London mortality

TABLE 3. SUMMARY OF KEY QUANTITATIVE CONCLUSIONS BASED ON NEWLY AVAILABLE EPIDEMIOLOGICAL STUDIES OR ANALYSES RELATING HEALTH EFFECTS TO ACUTE EXPOSURE TO AMBIENT AIR LEVELS OF SO_2 AND/OR PM

Type of Study	Results Obtained	24-hr average pollutant level (μg/m³)			Reference
		BS*	TSP	SO_2	
Mortality	Indications of increased mortality during London winters of 1958-59 to 1971-72, with most marked SO_2 effects evident at ~700-750 μg/m³ and indications of small increases at BS levels <500 μg/m³ and possibly as low as 150-300 μg/m³.	<500	--	~700-750	Mazumdar et al. (1982)
	New analyses of same 1958-59 to 1971-72 London winter mortality data indicative of increased mortality at BS levels <500 μg/m³ and no evident threshold at 150 μg/m³.	<150-500	--	--	Ostro (1984)
	Unpublished reanalysis of same 14 year London mortality data using spectral transform multiple regression analyses confirming significant associations for total, cardiovascular and respiratory mortality, accounting for autocorrelation and temperature. Suggestion of more pronounced effects with 7-21 day cycles of exposure.	<500	--	--	Shumway et al. (1983)
	Unpublished reanalysis of same 14 year London mortality data using regression analyses that detrended data for time series autocorrelation, humidity, and temperature indicating significant associations between mortality and BS to below 100 μg/m³, but not for SO_2 at <500 μg/m³.	Continuous association from lowest (<100 μg/m³) BS levels	--	<500	Schwartz and Marcus (1986)
Morbidity	Evidence for reversible (~2-3 wk) small (2-3%), but statistically significant decrements in FVC of school children following episodes in Steubenville, Ohio when 24-h TSP and SO_2 levels respectively ranged up to 220-420 and 280-460 μg/m³, but not after "sham" episode with TSP = 160 and SO_2 = 190 μg/m³. Larger decrements seen in subset of children.	--	220-420	280-460	Dockery et al. (1982)
	Evidence for reversible (~2-3 wk) small (3-5%), but statistically significant decrements in pulmonary function measures (FVC, $FEV_{1.0}$, MEF) for school children in the Netherlands during and after pollution episode when 24-h TSP, RSP, and SO_2 levels ranged up to 200-250 μg/m³, but no effect shortly after day when same pollutants averaged 100-150 μg/m³.	--	200-250	200-250	Dassen et al. (1986)

*Note that it is impossible to quantify with any precision PM levels (in μg/m³) associated with increased mortality risks at low BS levels, due to lack of gravimetric calibrations for BS readings beyond 1963 and difficulties in ruling out contributions potentially due to other factors.

experience: (1) Markedly increased mortality occurred, mainly among the elderly and chronically ill, in association with BS and SO_2 concentrations above 1000 $\mu g/m^3$, especially during episodes when such pollutant elevations occurred for several consecutive days; (2) During such episodes coincident high humidity or fog was also likely important, possibly by providing conditions leading to formation of H_2SO_4 or other acidic aerosols; (3) Increased risk of mortality is associated with exposure to BS and SO_2 levels in the range of 500 to 1000 $\mu g/m^3$, for SO_2 most clearly at concentrations in excess of ~700 $\mu g/m^3$; and (4) Convincing evidence indicates that relatively small but statistically significant increases in the risk of mortality exist at BS (but not SO_2) levels below 500 $\mu g/m^3$, with no indications of any specific threshold level having been demonstrated at lower concentrations of BS (e.g., at $\leq$150 $\mu g/m^3$). However, precise quantitative specification of the lower PM levels associated with mortality is not possible, nor can one rule out potential contributions of other possible confounding variables at these low PM levels.

Besides the above London mortality analyses, additional studies reviewed in this chapter evaluated relationships between mortality and short-term PM/SO_x exposures in various other geographic locations. For example, newly available reanalyses of New York City data by Ozkaynak and Spengler (1985) raise possibilities that, with additional work, further insights may emerge regarding mortality-air pollution relationships in a large U.S. urban area. However, the interim results reported thus far do not now permit definitive determination of their usefulness for defining exposure-effect relationships, given the above-noted types of caveats and limitations. Similarly, it is presently difficult to accept the findings reported by Hazakis et al. (1986) of mortality associated with relatively low levels of SO_2 pollution in Athens, given questions stated above regarding representativeness of the monitoring data and the statistical soundness of using deviations of mortality from an earlier baseline relatively distant in time. Lastly, newly reported analyses of mortality-air pollution relationships in Pittsburgh (Allegheny County, PA) reported by Mazumdar and Sussman (1983) utilized inadequate exposure characterization and the results contain sufficient internal inconsistencies, so that the analyses are not useful for delineating mortality relationships with either SO_2 or PM.

As for newly-reported analyses of short-term PM/SO_x exposure-morbidity relationships discussed in this chapter, the Dockery et al. (1982) study noted in Table 3 provides the best-substantiated and most readily interpretable results. Those results, point toward decrements in lung function occurring in association with acute, short-term increases in PM and SO_2 air pollution. The small, reversible decrements appear to persist for up to 2-3 wks after episodic exposures to these pollutants across a wide range, with no clear delineation of threshold yet being evident. In some study periods effects may have been due to 24-hr TSP and SO_2 levels ranging up to 422 and 455 $\mu g/m^3$, respectively. Notably larger decrements in lung function were discernable for a subset of children (responders) than for others. The precise medical significance of the observed decrements per se or any consequent long-term sequalae remain to be determined. The nature and magnitude of lung function decrements found by Dockery et al. (1982) are also consistent with: (1) the recently reported findings of the Dassen et al. (1986) study noted in Table 3 for Dutch children; (2) observations of Stebbings and Fogleman (1979) of gradual recovery in lung function of children during seven days following a high PM episode in Pittsburgh, PA (max 1-hr TSP estimated at 700 $\mu g/m^3$); (3) and the report of Saric et al. (1981) of 5 percent average declines in $FEV_{1.0}$ being associated with high SO_2 days (89-235 $\mu g/m^3$).

Table 4 summarizes those newly available epidemiology studies which appear to provide the most useful quantitative evidence for morbidity effects associated with long-term (generally annual-average) exposure to PM and/or SO_2. Note that, as was the case for the earlier criteria review (U.S. EPA, 1982a), none of the newly available analyses of relationships between mortality and chronic PM and/or SO_x exposures were judged here to yield sufficiently quantitative information to be useful for derivation of criteria.

From among the numerous new studies published on morbidity effects associated with long-term exposures to PM or SO_x, only the few listed in Table 4 are judged here to provide potentially useful results by which to derive quantitative conclusions concerning exposure-effect relationships for the subject pollutants. The Ware et al. (1986) study provides evidence of respiratory symptoms in children being associated with particulate matter exposures in contemporary U.S. cities without evident threshold across a range of TSP levels for ~30 to 150 $\mu g/m^3$. The increase in symptoms appears to occur without concomitant decrements in lung function among the same children. The medical significance of the observed increases in symptoms unaccompanied by decrements in lung function reremains to be fully evaluated but is of likely health concern. Caution is

TABLE 4. SUMMARY OF KEY QUANTITATIVE CONCLUSIONS BASED ON NEWLY AVAILABLE EPIDEMIOLOGICAL STUDIES RELATING HUMAN HEALTH EFFECTS TO LONG-TERM EXPOSURES OF SO_2 AND/OR PM

		Annual-Average pollutant levels ($\mu g/m^3$)		
Type of Study	Results Obtained	TSP	SO_2	Reference
Initial Cross-Sectional Analyses of Ongoing Longitudinal Study	Increased rates of cough, bronchitis and lower respiratory disease (in the absence of lung function changes) among school children in 6 U.S. cities significantly associated with annual-average TSP levels across range of approximately 30 to 150 $\mu g/m^3$ when analyzed for between city effects but not in relation to PM gradients within individual cities. Effects most clear for highest PM areas (~60-150 $\mu g/m^3$) versus lowest (~40-60 $\mu g/m^3$). No significant association with SO_2 except for cough.	~60-150	--	Ware et al. (1986)
Cross-Sectional Study	Significantly increased rates of persistent cough and phlegm (PCP) among young adults associated with annual-average $SO_2 \simeq 115\ \mu g/m^3$ in highest exposure Utah community versus 3 lower exposure towns with SO_2 in 11-36 $\mu g/m^3$ range. No TSP gradient across four communities. Effects possibly due to intermittant high SO_2 peaks.	--	115	Chapman et al. (1985)
Longitudinal Study in Southwestern U.S. towns	Significantly increased prevalence of cough among children from highest pollution area (annual average $SO_2 = 103\ \mu g/m^3$; intermittant 3 h peaks often exceeded 2,500 $\mu g/m^3$ or ~1 ppm) in comparison to lower pollution towns (annual $SO_2 = 14\ \mu g/m^3$). No TSP gradient across high and low pollution towns. Effects possibly due to intermittant high SO_2 peaks.	--	103	Dodge et al. (1985)

warranted, however, in using these findings for risk assessment purposes in view of the lack of significant associations for the same variables when assessed from data within individual cities included in the Ware et al. (1986) study. The findings derived from the series of studies by Ostro (Ostro, 1983; Hausman et al., 1984; Ostro, 1987), also discussed in the present chapter, are qualitatively indicative of morbidity effects in adults being associated with PM exposures over time within U.S. cities and tend to support the plausibility that the associations observed by Ware et al, (1986) reflect actual morbidity effects in children due to contemporaneous U.S. PM and SO_2 exposures.

The other new American studies listed in table 4 provide evidence for: (1) increased respiratory symptoms among young adults in association with annual-average SO_2 levels of $\sim$115 $\mu g/m^3$ (Chapman et al., 1985); and (2) increased prevalence of cough in children (but not lung function changes) being associated with intermittent exposures to mean peak 3-hr SO_2 levels of $\sim$1.0 ppm or annual-average levels of $\sim$103 $\mu g/m^3$ (Dodge et al., 1985). It is difficult to determine if effects observed in these two studies are due to repeated high-level short-term SO_2 peak exposures or to more chronic exposure to lower annual-average levels of SO_2 or its transformation products.

Results from one European study (PAARC, 1982a,b) also tend to suggest that increased lower respiratory disease symptoms and decrements in lung function in adults (both male and female) may be associated with annual average SO_2 levels increasing across a range from about 25 to 130 $\mu g/m^3$. In addition that study suggests that upper respiratory disease and lung function decrements in children may also be associated with annual-average SO_2 levels across the above range. However, the SO_2-morbidity effects associations reported by PAARC (1982a,b) cannot be fully accepted in view of (1) internal inconsistencies between findings obtained with SO_2 exposure estimates based on one type of measurement method versus those based on another measurement technique, and (2) the lack of adequate control for potentially important confounding factors in certain of the analyses yielding significant associations.

CHAPTER 4. CONTROLLED HUMAN EXPOSURE STUDIES OF SULFUR DIOXIDE HEALTH EFFECTS

Since the completion of the 1982 EPA criteria document (U.S. EPA, 1982a) and the first addendum to it (U.S. EPA, 1982c), numerous scientific articles have been published in the peer-reviewed literature or accepted for publication in regard to controlled human exposure studies providing important additional information pertinent to development of criteria for primary (health related) NAAQS for SO_2. This chapter of the present addendum summarizes and evaluates the newly available studies and discusses their relationship with certain other key studies and conclusions from Chapter 13 of the 1982 criteria document and the earlier addendum. Several of the key issues discussed in the previous addendum have been further investigated. Those discussed here are:

(1) Differences in subject characteristics, medication, and restriction from medication which may have considerable impact upon the differences in results reported by different laboratories.

(2) Concentration (SO_2)-response relationships in sensitive individuals under various conditions of exercise activity level or other form of hyperpnea.

(3) Possible enhancement of SO_2-induced bronchoconstriction by cold and/or dry air and by mouthpiece breathing.

(4) Mechanisms of action of SO_2-induced bronchoconstriction in sensitive (asthmatic) individuals.

The majority of subjects used in the studies summarized in this addendum were asthmatic. Asthma is a heterogeneous disease classification which includes a broad range of subjects. The least severe asthmatic may have had asthma diagnosed by a physician during childhood (by an unknown set of criteria) and have been mainly symptom-free since childhood and rarely, if ever, requires medication. On the other end of the spectrum are individuals who may be on chronic bronchodilator therapy (theophylline), who may use chromolyn (disodium chromoglycate) prior to activity, and may also require steroids. Pulmonary function tests (spirometry and airway resistance) are used to define the clinical status of an asthmatic at the time the studies are performed. Since airway obstruction in asthma is variable and often intermittent, and given that the physiologic status is highly influenced by the quantity and type of medication being used, tests of lung function cannot be used alone to determine the severity of the disease at any one time.

In addition to the diversity of clinical status, there was a broad range of selection criteria used to define asthma in various laboratories and from study to study. In some of the early studies, a clinical definition of asthma (i.e., diagnosed by a physician) was the selection criterion. In an effort to provide more descriptive information about the subjects, other criteria such as a positive response (i.e, much more reactive than "normal" subjects) to a pharmacologic stimulus such as methacholine or histamine was used as a criterion for selection. A positive (bronchoconstriction) response to an exercise test (5 to 10 min at 85 percent of maximum) or to an SO_2 inhalation challenge was also used to select subjects. The use of these descriptive criteria is sometimes useful in comparing results among laboratories.

One further point which relates to severity of asthma is the ability of the subjects to safely withhold their medication for a particular period of time. There was considerable variation among laboratories in the duration of time for which certain types or general classes of medication were restricted. A number of the characteristics of the subjects who participated in studies described in this addendum are summarized in Tables 5 and 6 along with other information on aspects of protocols employed in the studies.

The criteria for adverse health effects of air pollutants have been a matter of considerable discussion and disparity of opinion. In general terms, the adversity or "clinical significance" of a response may be discussed in relation to the magnitude of the functional changes (this must be considered on a test-specific basis), the duration or persistence of the response (i.e., acute responses vs. permanent or long-term health effects) the types of symptoms and the degree of discomfort or distress involved, and also the need for possible therapeutic intervention.

The relationship of symptoms and measured physiological responses to the health status of asthmatics may not always be readily apparent. Figure 7 may

TABLE 5. SUMMARY OF ASTHMATIC SUBJECT CHARACTERISTICS FROM NEWLY AVAILABLE CONTROLLED HUMAN EXPOSURE STUDIES OF EFFECTS OF SULFUR DIOXIDE ON PULMONARY FUNCTION

Reference	Age	Number of Subjects M/F	Baseline[5] SRaw $\bar{x}$ (Range)[1]	Baseline[5] $FEV_{1.0}$/FVC% $\bar{x}$ (Range)[1]	Major Medications	Medication Exclusions	Medication[3] Withheld	Allergy History	PRELIMINARY CHALLENGES: Reversibility of Obstruction	PRELIMINARY CHALLENGES: Histamine Methacholine	PRELIMINARY CHALLENGES: SO_2	PRELIMINARY CHALLENGES: Exercise Induced Bronchoconstriction
Bethel et al. (1983a)	22-36	8/2	5.0 (3.4-7.2)	79(63-89)	---[4]		48 h	Y	Y	HIS	---	---
Bethel et al. (1983b)	20-37	3/6	4.4 (1.7-7.2)	82(73-93)	---[4]		48 h	Y	Y	HIS	Y	---
Bethel et al. (1984)	25-36	5/2	6.6 (3.4-13.8)	71(52-84)	---[4]		48 h	Y	Y	HIS	Y	---
Bethel et al. (1985)	22-46	16/3	6.0 (3.1-10.6)	77(58-89)	---	---	12 h	MOST	Y	HIS	---	---
Hackney et al. (1984)	$\bar{x}$=25±4	13/4	5.94 (±4.03 S.D.)	76(±13 S.D.)	Half on Broncho dilators	No Cromolyn No Steroids	study day 12 h	---	---	---	Y[6]	---
Kehrl et al. (1986)	20-30	10/0	4.8 (2.0-8.5)	76(61-87)	Broncho dilators	No Cromolyn No Steroids	48 h oral 12 h inhal.	TESTED[6]	---	MET	Y[6]	---
Koenig et al. (1983b)	12-16	6/3	---	---	Theophylline	---	4 h	Y	Y	---	---	Y
Koenig et al. (1985)	14-18	5/5	---	---	Theophyl. Albuterol	---	4 h	Y	Y	---	---	?
Linn et al. (1982)	18-30 $\bar{x}$=23	13/11	4.3 (NA)	80(±10 S.D.) (lowest 59)	None	No Routine Medication	1 week	Y	---	MET	---	---
Linn et al. (1983a)	18-30 $\bar{x}$=23	15/8	4.9 (±2.0 S.D.)	78(±11 S.D.)	None	No Routine Medication	1 week	Y	---	---	?	---
Linn et al. (1983b)	19-31	14/10	5.2 (1.7-17.5)	76(58-100)	---	---	12 h	---	---	MET[7]	?	Y

continued on following page

TABLE 5. SUMMARY OF ASTHMATIC SUBJECT CHARACTERISTICS FROM NEWLY AVAILABLE CONTROLLED HUMAN EXPOSURE STUDIES OF EFFECTS OF SULFUR DIOXIDE ON PULMONARY FUNCTION (continued)

									PRELIMINARY CHALLENGES			
Reference	Age	Number of Subjects M/F	Baseline[5] SRaw $\bar{x}$ (Range)[1]	Baseline[5] $FEV_{1.0}$/ FVC% $\bar{x}$ (Range)[1]	Major Medications	Medication[2] Exclusions	Medication[3] Withheld	Allergy History	Reversi-bility of Obstruc-tion	Hista-mine Metha-choline	SO_2	Exercise Induced Bronchocon-striction
Linn et al. (1984a)	19-30	19/7	5.2 (1.5-17.5)	76(50-90)	---	---	12 h	---	---	MET[7]	Y	Y
Linn et al. (1984b)	19-33	13/11	5.0 (2.9-8.3)	81(67-100)	5 Broncho Dilator	---	8 h inhaled 12 h oral 48 h antihist.	---	---	---	Y	Y
Linn et al. (1984c)	18-33	12/2	5.2 (1.8-7.5)	75(51-87)	Occasional Bronchodil.	---	12 h	---	---	---	Y[4]	Y
Linn et al. (1985a)	19-33	13/9	5.0- (2.8-8.2)	80(67-100)	Occasional Bronchodil.	---	8 h inhaled 12 h oral 48 h antihist	---	---	---	Y[4]	Y
Linn et al. (1985b) (COPD)	49-68 $\bar{x}$=60	15/9	19.0 (NA)	47(27-70)	Regular Bronchodil.	---	4 h	---	---	---	---	---
Roger et al. (1985)	19-34	28/0	6.7 (2.0-12.8)	73(56-89)	Broncho dilators	No Cromolyn No Steroid	12 h inhaled 48 h oral	Y[6]	---	MET	---	---
Schachter et al. (1984)	$\bar{x}$=27±5	4/6	5.0 (±1.5)	NA	Broncho dilators	---	24 h	Y	---	MET[8]	---	---
Sheppard et al. (1983)	22-36	4/4	7.6 (3.2-14.9)	75(51-91)		---	8 h inhaled 12 h oral 48 h antihist.	---	Y	HIST	Y	---
Sheppard et al. (1984)	20-37	5/3	8.3 (4.0-18.5)	68(52-85)	Metaproterenol Albuterol Theophylline	---	48 h antihist. 24 h theophyl. 10 h sympath	---	Y	---	---	---
Snashall and Baldwin (1982)	25-61	4	7.3	---	2 Cromolyn 1 Steroid	---	24 h var.	Y	---	---	---	---

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TABLE 5. SUMMARY OF ASTHMATIC SUBJECT CHARACTERISTICS FROM NEWLY AVAILABLE CONTROLLED HUMAN EXPOSURE STUDIES OF EFFECTS OF SULFUR DIOXIDE ON PULMONARY FUNCTION (continued)

									PRELIMINARY CHALLENGES			
Reference	Age	Number of Subjects M/F	Baseline[5] SRaw $\bar{x}$ (Range)[1]	Baseline[5] $FEV_{1.0}$/FVC% $\bar{x}$ (Range)[1]	Major Medications	Medication[2] Exclusions	Medication[3] Withheld	Allergy History	Reversibility of Obstruction	Histamine Methacholine	SO_2	Exercise Induced Bronchoconstriction
Koenig et al. (1987)	13-17	3/7	---	---	None	---	---	Y	---	---	---	Y
Myers et al. (1986a)	19-40	7/3	---	75.5 (61-92)	1 Theophylline	No cromolyn No steroids	8 h inhaled 12 h theophylline	Y	Y	MET/HIST	Y[9]	---
Myers et al. (1986b)	19-40	9/2	---	81.5 (61-89)	---	---	8 h inhaled 18 h theophylline	Y	Y	MET/HIST	---	---
Tan et al. (1982)	16-60	14/8	0.92 KPA/1/s	---	2 Cromolyn Symp-Bronch	---	12 h bronch 24 h chrom or steroid 72 h antihist.	Y	Y	---	---	---

[1] Range except ± SD where indicated.

[2] Subjects were excluded from the study if using this medication.

[3] Number of hours medication withheld before start of exposure.

[4] No chronic bronchodilator therapy (theophylline).

[5] Baseline SRaw and $FEV_{1.0}$/FVC ratio were taken from the subject characterization when available, otherwise from pre-exposure measurements.

[6] Subjects were given allergy skin tests.

[7] Most but not all subjects were tested - most of these were reactive.

[8] Methacholine reactivity determined afterward.

[9] Selection of subjects based on previously determined SO_2 response.

TABLE 6. SUMMARY OF NORMAL SUBJECT CHARACTERISTICS FROM NEWLY AVAILABLE CONTROLLED HUMAN EXPOSURE STUDIES OF EFFECTS OF SULFUR DIOXIDE ON PULMONARY FUNCTION

Author	Age	N M/F	Baseline SR_{aw}	Baseline $FEV_{1.0}/FVC$	Allergy History	Challenges
Bedi et al. (1984)	19-28	23/0	$\bar{x}$ = 7.41	0.70-0.96	---	---
Folinsbee et al. (1985)	19-28	22/0	$\bar{x}$ = 7.63	$\bar{x}$ = .81	---	---
Kulle et al. (1984)	21-34	10/10	$\bar{x}$ = 5.52	$\bar{x}$ = .83	N	MET
Stacy et al. (1983)	(18-40)	231/0	4.77*	0.83*	N	---
Schachter et al. (1984)	26.1	5/5	---	---	---	---
Rondinelli et al. (1986)	55-73	10/0	---	$\bar{x}$ = .84	N	MET

*Subset of the data.

GRADE OF RESPONSE	NONE	MILD	MODERATE	SEVERE	INCAPACI-TATING
CHANGE IN SRAW	NO CHANGE	INCREASE LESS THAN 100%	INCREASE UP TO 200%•	INCREASES MORE THAN 200%•	INCREASES >>200%•
DURATION OF EFFECT	NA	SPONTANEOUS RECOVERY <30 MIN	SPONTANEOUS RECOVERY <1 HR	BRONCHODILATOR REQUIRED TO RESOLVE SYMPTOMS	EMERGENCY TREATMENT REQUIRED
CHANGE IN SPIROMETRY $FEV_{1.0}$, FVC	NO CHANGE	NO CHANGE	DECREASE BUT <15%•	DECREASE >15%•	DECREASE >>15%•
SYMPTOMS	NO RESPIRATORY SYMPTOMS	MILD SYMPTOMS NO WHEEZE OR CHEST TIGHTNESS	SOME WHEEZE• OR CHEST TIGHTNESS	OBVIOUS WHEEZE• MARKED CHEST TIGHTNESS BREATHING DISTRESS	SEVERE BREATHING DISTRESS•

• STATISTICALLY SIGNIFICANT CHANGE

Figure 7. Gradation of physiological responses to SO_2.

be used to roughly classify the severity of response using four variables which are frequently measured in the studies discussed in this chapter, namely: (a) change in SRaw; (b) duration of effect of SO_2; (c) changes in spirometry, chiefly $FEV_{1.0}$; (d) types of symptoms and relative discomfort. This table is not intended to provide a quantitative description of what does or does not constitute an adverse health effect but is primarily intended to demonstrate that there are an array of responses and to assist the reader in judging the relative severity of the different responses which are described. There is no question that the types of response described under INCAPACITATING would be considered as clinically significant adverse health effects. Most of the responses identified in this chapter would fall in the moderate and severe categories or some combination of the responses described under those categories.

4.1. NORMAL SUBJECTS EXPOSED TO SULFUR DIOXIDE

The pulmonary function effects of SO_2 in normal healthy adult volunteers have usually been much less than those seen in SO_2-exposed subjects with clinically documented asthma. The newly available information supports this conclusion in general but also suggests that some mild effects which are of little if any acute health importance may be observed in normal subjects at concentrations below 5.0 ppm. The 1982 criteria document (U.S. EPA, 1982a) presented the conclusion that the probable lowest-observable-effects level in normal healthy subjects is 5.0 ppm SO_2 at rest. The first addendum to that document (U.S. EPA, 1982c) further suggested that normal subjects are about one order of magnitude (i.e., tenfold) less sensitive to SO_2 exposure than asthmatics.

Bedi et al (1984) studied subjects exposed to 1.0 and 2.0 ppm SO_2 in an environmental chamber (22°C, 40 percent RH) for 2h ($\dot{V}_E$ = 40 L/min for 3 to 30 min exercise periods with intervening 10 min rest). In the initial 9 subjects tested at both 1.0 ppm and 2.0 ppm SO_2, these investigators reported a modest (10.3 percent) but significant increase in SRaw following both exposure concentrations. Further investigation with a total of 22 subjects at 1.0 ppm using the same protocol failed to substantiate this finding. Given the trivial increase in SRaw (well within daily variations), the finding in the initial group probably occurred by chance. Folinsbee et al. (1985) also reported exposure of normal subjects to 1.0 ppm SO_2 in a study in which the effects of combined exposure to ozone and SO_2 were examined. The exposure protocol for this study was the same as the Bedi et al. (1984) study and included many of the same subjects. There were no significant changes in forced expiratory spirometry or airway resistance as a result of 1.0 ppm SO_2 exposure reported for these subjects.

Stacy et al. (1983) exposed subjects to 0.75 ppm SO_2 alone and in combination with several particulate pollutants. During the 4-h exposures, subjects walked on a treadmill on two occasions ($\dot{V}_E$ approximately 55 L/min). There were no significant effects of this SO_2 (or SO_2 plus particulate) exposure on either forced expiratory spirometry or airway resistance.

Schachter et al. (1984) compared the responses of asthmatics and normals (4M, 6F) to SO_2. Three of the normals were reportedly atopic (i.e., they probably had some history of allergy). There were no significant effects in normal subjects at any of the concentrations tested (0.25, 0.50, 0.75, and 1.0 ppm SO_2). Measurements were made for 60 min following a 10-min bicycle exercise period ($\dot{V}_E$ estimated at 35 L/min by measurement at the same workload on another occasion) in SO_2; the SO_2 level was maintained for the first 30 min post-exercise. At the higher SO_2 concentrations (0.75 and 1.0 ppm) the subjects did experience upper respiratory symptoms (these included unpleasant taste and odor and sore throat, symptoms associated with extrathoracic airways).

Koenig and Pierson (1985) in a review of several studies from their laboratory reported a decline (6 percent) in $FEV_{1.0}$ following exposure to 1.0 ppm SO_2 in 8 healthy normal adolescents. These subjects were exposed via mouthpiece to either 1 ppm SO_2, 1 mg/m^3 NaCl aerosol, or their combination. Resting exposure of 30 min was followed by 10 min of exercise ($\dot{V}_E$ = 39.9 L/min). The apparent decrease in $FEV_{1.0}$ occurred 2 to 3 min following the exercise period in SO_2. However, the $FEV_{1.0}$ decrease following saline aerosol was 4 percent and the absolute post-exposure $FEV_{1.0}$ values were identical (i.e., 2.89 liters). Furthermore, the authors used repeated pair t-tests in their analysis without correction for multiple comparisons (e.g., Bonferroni). These data should be subjected to a more rigorous statistical analysis to ascertain their significance. Even if these $FEV_{1.0}$ data were statistically significant, the differences between the air exposure and SO_2 exposure are so small that they are of no clinical importance.

Exposure to a mixture of SO_2 (1 ppm) and ammonium sulfate (528 μg/m^3) was studied in 20 normal subjects by Kulle and associates (1984). The subjects were young adult nonsmokers (10M, 10F) with normal spirometry and no allergic or respiratory disease history. Four hour exposures occurred in an environmental chamber (22°C, 60 percent RH) and included two 15-min exercise periods (mild-100 watts, $\dot{V}_E$ estimated 40 L/min [4 to 5 times rest]). There were no significant effects on spirometry or airway resistance after exposure to either SO_2 alone, ammonium sulfate alone, or their combination. There was no change in the response to a methacholine inhalation challenge following any of the exposures. There were reports of upper respiratory symptoms which were most prevalent with the combination exposure. This study further supports the absence of pulmonary function effects of SO_2 at 1.0 ppm in normal subjects.

Wolff et al. (1984) exposed nine steel workers, two of whom were classified as asthmatic, to 5 ppm SO_2 or SO_2 plus carbon dust for 2.5 h in an environmental chamber (22°C, 50 percent RH). The exposure included five 4-min exercise periods ($\dot{V}_E$ not reported). Mucociliary clearance measurement exhibited no consistent pattern of change. Histamine reactivity (percent drop in $FEV_{1.0}$ at threshold dose) showed a tendency to increase slightly (37 percent; 28 percent excluding asthmatics). There were no notable changes in pulmonary function among the normal subjects. Symptomatically the subjects found the SO_2 plus carbon dust exposure more unpleasant than SO_2 alone.

The effects of SO_2 on ten older men (55 to 73) were studied by Rondinelli and colleagues (1986). Subjects were exposed via mouthpiece at rest (10 ℓ/min) and exercise (10 min at 31 ℓ/min) first to NaCl droplet aerosol, and then to either NaCl aerosol plus 0.5 ppm SO_2 (n=7), or NaCl aerosol plus 1.0 ppm SO_2. $FEV_{1.0}$ decreased after exercise in all conditions by 5, 7, and 8 percent respectively. Although these results are suggestive of a small effect of oral breathing of SO_2 in older men, the incompletely randomized exposure sequence and the inappropriate use of repeated paired t-tests in the analysis raise sufficient questions that the effects cannot be considered conclusive.

In summary, these studies of SO_2 exposure in normal healthy adults and adolescents demonstrate minimal, if any, significant pulmonary function effects of SO_2 exposure at 0.25 to 2.0 ppm with exposure durations ranging from 10 minutes to four hours including exercise periods, with work outputs sufficient to increase ventilation to 35 to 55 L/min. The only effect of any consequence was the increase in upper respiratory symptoms, which was chiefly the result of the unpleasant taste/odor of sulfur dioxide.

4.2 CHRONIC OBSTRUCTIVE PULMONARY DISEASE PATIENTS EXPOSED TO SO_2

In addition to asthmatics, patients with chronic obstructive pulmonary disease (COPD) have also been exposed to SO_2. Linn et al. (1985b) exposed 24 COPD patients (ages 49 to 68) to 0.4 and 0.8 ppm SO_2. Although there was a wide range of functional impairment ($FEV_{1.0}/FVC$ ratio ranged from 27 to 70 percent), all patients were able to exercise without supplemental oxygen. One-hour exposures in an environmental chamber (22.5°C, 86 percent RH) included two 15-min exercise periods ($\dot{V}_E$ = 18 L/min). In contrast to many previous studies of mild asthmatics, most of these patients regularly used bronchodilators and were permitted their use up to 4 h prior to study. There were no effects of SO_2 exposure in this subject group and no trends indicative of change in any of the measured functions (including SRaw, spirometry, and arterial oxygen saturation). It should be noted that little if any effect would be anticipated in asthmatics under these exposure conditions. The authors suggested that these COPD patients may be less reactive to SO_2 than younger asthmatics, although, as the authors discuss, given the low dose rate of exposure and the marked differences in medication status, this conclusion may be premature. The ventilations achievable by COPD patients are limited by the severity of their disease. Further investigations of COPD patients exposed to SO_2 should include groups with less severe disease who are capable of exercising at moderate intensity (e.g., $\dot{V}_E$ = 30 to 35 ℓ/min) and able to withhold medication. Only after such investigations have been completed will sufficient information be available to assess the relative risk of COPD patients exposed to SO_2.

4.3 FACTORS AFFECTING THE PULMONARY RESPONSE TO SO_2 EXPOSURE IN ASTHMATICS

4.3.1 Dose-Response Relationships

Important considerations in assessing the response to any inhaled gas or aerosol include the concentration of the substance in the inspired air, the rate of exchange of ambient air with the lung (ventilation), and the duration of exposure. The concentrations to which asthmatics have been exposed in more recent studies (since 1981) range from 0.10 to 2.0 ppm SO_2 although interest has focused on the range from 0.2 to 1.0 ppm. A broad range of exposure durations has been utilized ranging from 3 min to 6 h, although the primary focus has been on 5 to 10-min exposures which incorporate hyperpnea. Ventilation rates have ranged from 8 to 10 L/min at rest to 60 to 70 L/min (exercise or voluntary eucapnic hyperpnea), although most interest has centered on moderate ($\dot{V}_E$ = 35 to 50 L/min) to heavy ($\dot{V}_E$ > 50 L/min) exercise levels which in warm humid environments provoke, at most, only mild to moderate exercise-induced bronchoconstriction. Results from the recently published studies are summarized in Table 7.

Schachter et al. (1984) performed a concentration-response study in a group of 10 normal subjects (see Section 4.1 above) and a group of 10 asthmatic subjects exposed in an environmental chamber (23°C, 70 percent RH) to 0, 0.25, 0.50, and 1.0 ppm SO_2. Subjects rested briefly and then exercised for 10 minutes at 450 kpm ($\dot{V}_E$ = 35 L/min). In addition, subjects were exposed to 1.0 ppm SO_2 at rest. A significant decline in $FEV_{1.0}$ followed both the 0.75 (-8.3 percent) and 1.0 (-13 percent) ppm exercise exposures in these asthmatics. This was accompanied by a significant increase (54 to 68 percent at 1.0 ppm) in airway resistance (interrupter method). There were also some changes (these did not occur consistently at all concentrations or time intervals after exposure) in maximum expiratory flow which mainly occurred at the two highest concentrations. The recovery was rapid and pulmonary function was within 5 percent of baseline (and no longer significantly different) by 10 min post-exercise even though SO_2 exposure continued at rest. As other investigators have reported, there was a considerable range of response among these subjects, with 3 or 4 subjects demonstrating no appreciable response to SO_2 at any concentration while some others showed trends indicative of a dose-response (SO_2-$FEV_{1.0}$) relationship beginning as early as 0.25 ppm. The responses of asthmatics seen in this study may appear less severe than those seen by other investigators at similar SO_2 concentrations, although comparisons are difficult because of the different measurements made; the relatively small changes in Raw may be partially due to the use of the interrupter method. However, a number of other factors could account for the discrepancies between this and other recent studies of asthmatics. First, the subjects were not pre-selected for the presence of airway hyperreactivity to SO_2, cold air, exercise, histamine or methacholine, an approach frequently used by others. Second, the moderate workload and unencumbered oronasal ventilation probably resulted in a lower SO_2 delivery to the reactive airways than would occur with mouth breathing.

TABLE 7. SUMMARY OF RESULTS FROM CONTROLLED HUMAN EXPOSURE STUDIES OF PULMONARY FUNCTION EFFECTS ASSOCIATED WITH EXPOSURE OF ASTHMATICS TO SO_2

Conc.	Duration	Number of Subjects*	Exposure Mode	Exposure Status	Observations	Comments	References
0.1 ppm	3 min.	8	Oral-mouthpiece 22°C 0% RH AH < 1	Hyperventilation to V_E = 51 l/min	Ventilation rate needed to increase SR_{aw} by 80% over resting baseline shifted by 3.8 l/min (7%) less than that needed for comparable HIB in dry air.	Symptom data not reported. Suggests marginal decrease in hyperventilation needed to produce HIB in dry air. Health significance unclear.	Sheppard et al. (1984)
0.2 ppm	5 min.	23	Chamber- 23°C 85% RH AH = 17.5	Exercising V_E = 48 l/min	No significant change in SR_{aw}, FEV_1, FVC, PEFR, $V_{max_{25-75}}$ over exercise control. Possibly statistically significant increase in overall symptom score but not for any one symptom.	No measureable physiologic changes with possible increase in symptom scores of uncertain significance.	Linn et al (1983b)
0.2 ppm	5 min.	8	Chamber- 5°C 1) 50% RH AH = 3.4 2) 85% RH AH = 5.8	Exercising V_E = 50 l/min	No significant changes in SR_{aw}, FEV_1, FVC, SG_{aw} over exercise control for either RH level. Suggestion of small increase in symptoms but no statistics given.	No measureable enhancement of SO_2 response for 5°C, 50% RH. Symptom score results of uncertain significance.	Linn et al. (1984a)
0.25 ppm	10 to 40 min.	10	Chamber- 23°C 70% RH AH = 14.4	Exercising V_E = 35 l/min	No significant changes in R_{aw}, FEV_1, MEF_{40}, with small (4%) change in $V_{max_{50}}$. No clear increase in symptoms, suggestion of increased response in 2 of 10 subjects.	Indicates no effect. Changes even in sensitive subjects of uncertain health significance.	Schachter et al. (1984)
0.25 ppm	5 min.	1) 19 2) 9	Chamber 23°C D.P. = 7.6°C (36% RH) AH = 7.4	Exercising 1) V_E = 60 l/min estimated (750 kpm-min) 2) V_E = 80-90 l/min estimated (1000 kpm-min)	With 750 kpm/min exercise, increase in SRaw in SO_2 (mean = 134%) signif. greater than clean air (mean = 77%). At 1000 kpm/min, no sig. diff. between SO_2 and clean air.	Effects at this level small or non-existent in comparison to heavy exercise alone. No symptoms reported. Response highly variable. Suggests 0.25 close to threshold for bronchoconstriction.	Bethel et al. (1985)
0.25 ppm	10 min. to 75 min.	28	Chamber 26°C 70% RH AH = 17.1	Intermittent exercise (3 10 minute periods) V_E = 42 l/min	No significant changes in SR_{aw}, TGV, resistance impedence for any of measurement periods. No significant changes in symptoms.	No measurable physiological or symptoms changes seen with .25 ppm SO_2 at this exercise level.	Roger et al. (1985)

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TABLE 7. SUMMARY OF RESULTS FROM CONTROLLED HUMAN EXPOSURE STUDIES OF PULMONARY FUNCTION EFFECTS ASSOCIATED WITH EXPOSURE OF ASTHMATICS TO SO_2 (continued)

Conc.	Duration	Number of Subjects	Exposure Mode	Exposure Status	Observations	Comments	References
0.25 ppm	3 min.	8	Oral-mouthpiece 22°C 0% RH AH = 1	Hyperventilation to V_E = 51 l/min	Ventilation needed to increase SR_{aw} by 80% over resting baseline shifted to 5.6 l/min (10%) less than that needed for comparable HIB in dry air.	Symptom data not reported. Suggests small decrease in exercise needed to produce HIB in dry air. Health significance unclear.	Sheppard et al. (1984)
0.3 ppm	5 min.	24	Chamber 80% RH 1) -6°C 2) 7°C 3) 21°C 1) AH = 2.5 2) AH = 6.2 3) AH = 14.7	Exercising V_E = 50 l/min	At -6°C, SR_{aw} increased 94% in air and 105% in SO_2. At 7°C SR_{aw} increased 59% in air and 87% in SO_2. At 21°C SR_{aw} increased 28% in air and 59% in SO_2. Increase in symptom scores at all temperatures slightly greater in SO_2 than in air.	Significant main effects at 0.3 ppm not reported. Symptom score changes generally mild and of uncertain significance to health. Under test conditions, results indicate SO_2 and moist cold air effects are additive or less than additive.	Linn et al. (1984b)
0.4 ppm	5 min.	23	Chamber 23°C 85% RH AH = 17.5	Exercising V_E = 48 l/min	Increased SRaw in SO_2 (69%) sig. diff. than increase in clean air (35%). Significant decrements in V_{max} (25-75) (mean=10%), but no significant changes in FEV_1. Significant increase in overall symptom score, but only one of 12 symptom categories increased significantly. One subject required medication to relieve distress.	Indicates moderate bronco-contriction. Overall symptom changes mild, but in at least one subject responses suggestive of clinical significance.	Linn et al. (1983b)
			5°C 1) 50% RH 2) 81% RH 1) AH = 3.4 2) AH = 5.8	V_E = 50 l/min	graphical depiction) and symptom score over exercise alone. Symptom score increase clearly larger for 50% RH than for 81% RH.	Significance of SG_{aw} and FEV_1 at 0.4 ppm not reported; indicates subjective response enhanced for dryer cool air even when measure of functional changes comparable to moist air	

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TABLE 7. SUMMARY OF RESULTS FROM CONTROLLED HUMAN EXPOSURE STUDIES OF PULMONARY FUNCTION EFFECTS ASSOCIATED WITH EXPOSURE OF ASTHMATICS TO SO_2 (continued)

Conc.	Duration	Number of Subjects	Exposure Mode	Exposure Status	Observations	Comments	References
0.4 ppm	5 min.	8	Chamber	Exercising	Apparent increase in SR_{aw} (from	No stat. for SR_{aw} changes.	Linn et al. (1984a)
0.5 ppm	10 to 40 min.	10	Chamber 23°C 70% RH AH = 14.4	Exercising V_E = 35 l/min	No significant changes in R_{aw}, FEV_1, MEF 40 with small (mean = 6%) decrement in V_{max50}. No clear increase in symptoms. Suggestions of increased FEV_1 response in 2 or 3 subjects.	Indicates minimal constriction for group at this exercise rate.	Schachter et al. (1984)
0.5 ppm	5 min.	10	Chamber 23°C 41% RH AH 8.4	Exercising V_E = 60 l/min estimated (750 kpm-min)	Increase in SRaw in SO_2 (mean = 238%) sig. diff. than increase in clean air (mean = 39%). Substantial variability in subjects; one showed eight-fold increase	Indicates substantial SO_2 induced bronchoconstriction at high exercise rate and mod. RH. No symptom data reported but extent of SR_{aw} changes suggestive of clinical significance.	Bethel et al. (1983a)
0.5 ppm	5 min.	9	80% RH, 23°C 1) Face mask 2) Mouthpiece AH = 16.5	Exercising 1) V_E = 27 l/min 2) V_E = 41 l/min 3) V_E = 61 l/min	Facemask exposure: No stat. sig. mean change in SR_{aw} with air or SO_2 at low or mod. exercise rate. For high exercise increase in SR_{aw} in SO_2 (219%) sig. larger than increase in clean air (25%) compared to mean baseline SR_{aw}. Percent ventilation breathed orally for the three exercise rates were: 1) 50%, 2) 52%, 3) 61%. Mouthpiece exposure: No sig. mean change in SR_{aw} for low exercise rate with moderate exercise, increased SR_{aw} in SO_2 (231%) sig. larger than clean air (5%). with high exercise, increased SR_{aw} in SO_2 (306%) sig. larger than clean air (25%)	Indicates SO_2 induced constriction enhanced by increased work rate, with protection afforded by oronasal (vs. oral) breathing greater at mod. than at high exercise rates. Asthmatics with rhinitis or other nasal blockage breathe more through mouth and appear at greater risk to SO_2 effects.	Bethel et al. (1983b)

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TABLE 7. SUMMARY OF RESULTS FROM CONTROLLED HUMAN EXPOSURE STUDIES OF PULMONARY FUNCTION EFFECTS ASSOCIATED WITH EXPOSURE OF ASTHMATICS TO SO_2 (continued)

Conc.	Duration	Number of Subjects	Exposure Mode	Exposure Status	Observations	Comments	References
0.5 ppm	30 min. rest 10 min. exercise	9	22°C 75+% RH AH = 14.6+	Mouthpiece 5-6 x rest V_E	Mouthpiece exposure: $FEV_{1.0}$ decreased, -15% (-4% in air); R_T increased 47%; $V_{max^{50}}$, $V_{max^{75}}$ decreased -30, -35%.	Indicates that mouthpiece breathing exacerbates the effect of SO_2 in asthmatics.	Koenig et al. (1983b)
		7		Facemask 5-6 x rest V_E	Facemask: No significant changes		
0.5 ppm	30 min. rest 20 min. exercise	10 (14-18 yr)	22°C 75% RH AH = 14.6	Mouthpiece 43 l/min exercise	Increase in nasal resistance of 32%, but not significant. FEV_1 decrease -24%, $V_{max^{50}}$ -46%; $V_{max^{75}}$ -56%. R_T increased 60%.	Indicates SO_2 may cause increased nasal resistance in asthmatics, which may result in more oral breathing and consequently more bronchoconstriction.	Koenig et al. (1985)
				Facemask	Significant increase in nasal resistance of 30%. FEV_1 decreased -16% $V_{max^{50}}$, $V_{max^{75}}$ -26%		
0.5 ppm	10 min. to 75 min.	28	Chamber 26°C 70% RH AH = 17.1	Intermittent exercise (3 10 min. periods) V_E = 42 l/min	Increased SR_{aw} in SO_2 (93%) sig. larger than clean air (47%). SR_{aw} increase after second and third exercise periods sig. less than after first ex. period. No significant changes in FVC, FEV_1, FEF Group mean symptoms for 20 subjects not sig. increased. Substantial variability in subjects, with one showing 11-fold increase in SR_{aw} and requiring medication to relieve pronounced symptoms.	Extent of effects are decreased after short-term repeated exercise. Broad degree of sensitivity to SO_2 with about 25% of subjects showing a 100% increase in SR_{aw}. Symptoms in at least one subject of clear clinical significance.	Roger et al. (1985)

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TABLE 7. SUMMARY OF RESULTS FROM CONTROLLED HUMAN EXPOSURE STUDIES OF PULMONARY FUNCTION EFFECTS ASSOCIATED WITH EXPOSURE OF ASTHMATICS TO SO_2 (continued)

Conc.	Duration	Number of Subjects	Exposure Mode	Exposure Status	Observations	Comments	References
0.5 ppm	3 min., repeated 3 times in succession at 30 min. intervals, again after 24 hrs and 1 week later	8	Oral-mouthpiece 23°C 82% RH AH = 16.9	Hyperventilation (varied for each subject)	Sig. increase in SR_{aw} ($\bar{x}$ = 104%) after first 3 min. exposure. After 30 min. rest, second response sig. but smaller (x = 35%); response after third exposure still smaller (x = 30%). SR_{aw} increase at 24 h. (x = 83%) and 1 week ($\bar{x}$ = 129%) not sig. diff. from increase after first 3 min. exposure.	Indicates repeated esposures to SO_2 can induce tolerance to bronchoconstrictive effects of SO_2 over a short period (>30 min) but not for longer periods.	Sheppard et al. (1983)
0.5 ppm	3 min.	7	Oral-mouthpiece 1) 23°C 77% RH 2) -11°C, "Dry" 1) AH = 15.8 2) AH < 1	Hyperventilation to "Threshold" V_E for each subject (30-50 l/min)	By design, increases in SR_{aw} or symptoms not sig. for SO_2 in warm, humidified air or cold dry air alone. Sig. increase in SR_{aw} ($\bar{x}$ = 222%) for combination of SO_2 and cold dry air. Six of seven subjects report wheezing and/or shortness of breath; two asked for medication. Symptoms not good indicator of of measured SR_{aw}.	Indicates that airway cooling, drying can increase SO_2 associated bronchoconstriction in hyperventilating asthmatics. Suggests synergism for these combinations.	Bethel et al. (1984)
0.6 ppm	5 min.	24	Chamber 80% RH 1) -6°C 2) 7°C 3) 21°C (1) AH = 2.5 (2) AH = 6.2 (3) AH = 14.7	Exercising V_E = 50 l/min	Increased SR_{aw} in SO_2 sig. greater than in clean air for all three temps. At -6°C, SR_{aw} increased 94% in air and 187% in SO_2. At 7°C, SR_{aw} increased 58% in air and 207% in SO_2. At 21°C, SR_{aw} increased 28% in air and 150% in SO_2. Symptom scores sig. greater in SO_2 than in air at all three temperatures.	Suggests that the bronchconstrictive effects of cold air and SO_2 combine in an additive or less-than-additive fashion. Some suggestion of cold air-SO_2 interaction in total asthma score. SR_{aw} changes suggestive of clinical significance at all temperatures.	Linn et al. (1984b)

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TABLE 7. SUMMARY OF RESULTS FROM CONTROLLED HUMAN EXPOSURE STUDIES OF PULMONARY FUNCTION EFFECTS ASSOCIATED WITH EXPOSURE OF ASTHMATICS TO SO_2 (continued)

Conc.	Duration	Number of Subjects	Exposure Mode	Exposure Status	Observations	Comments	References
0.6 ppm	5 min.	22	Chamber 21°C, 38°C 20% RH, 80% RH AH = 3.7, 14.7 @ 21°C AH = 9.3, 37.0 @ 38°C	Exercise V_E = 50 l/min	SR_{aw} changes in clean air ranged from -4% to +12%. With SO_2, at 21°C SR_{aw} increased 206% with dry and 157% with humid air, while at 38°C SR_{aw} increased 89% in dry air and 39% in humid air.	Indicates the importance of airway drying as an exacerbating factor in the induction of SO_2-bronchoconstriction.	Linn et al. (1985a)
0.6 ppm	Total 6 hr 2 successive days 2x5 min exer. each day, separated by 5h	14 (18-33)	Chamber 22°C 85% RH AH = 16.5	Exercise 50 l/min	After correction for clean air EIB, SR_{aw} increased 136, 120, 147, 100% on the early-day 1, late-day 1, early-day 2, late-day 2. No difference between times or days.	Indicates that refractory period for SO_2-induced bronchoconstriction is less than 5h.	Linn et al. (1984c)
0.6 ppm	5 min.	24	Chamber- 85% RH 1) 5°C 2) 22°C 1) AH = 3.4 2) AH = 16.5	Exercising V_E = 50 l/min	At 5°C, increased SR_{aw} with SO_2 (182%) sig. greater than clean air (38%). At 22°C, increased SR_{aw} with SO_2 (132%) sig. greater than clean air (27%). Lower respiratory and total symptom scores much greater in SO_2 than in clean air.	Suggests bronchoconstrictive effects of cold, moist air may increase SO_2 effects, but under these conditions, enhancement is inconsistent and not significant). Symptoms, SR_{aw} changes suggestive of clinical significance at both temperatures.	Linn et al. (1984a)
0.6 ppm	5 min.	8	Chamber- 5°C 1) 50% RH 2) 81% RH 1) AH = 3.4 2) AH = 5.8	Exercising V_E = 50 l/min Pilot study	Significant increase in SR_{aw} and symptom scores over exercise alone for both humidities (graphical depiction). No sig. diff. between humidities at this temperature.	Suggests that under these conditions, SO_2 response apparently not enhanced by lower humidity of cool air which has a low water content already.	Linn et al. (1984a)

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TABLE 7. SUMMARY OF RESULTS FROM CONTROLLED HUMAN EXPOSURE STUDIES OF PULMONARY FUNCTION EFFECTS ASSOCIATED WITH EXPOSURE OF ASTHMATICS TO SO_2 (continued)

Conc.	Duration	Number of Subjects	Exposure Mode	Exposure Status	Observations	Comments	References
0.6 ppm	5 min.	23	Chamber- 23°C 85% RH AH = 17.5	Exercising V_E = 48 l/min	Increased SR_{aw} in SO_2 (120%) sig. greater than in air (36%). Significant decline in FVC (mean = 3%), FEV_1 (mean = -13%), PEFR (mean = -26%) $V_{max25-75}$ x = -26%). Sig. increase in total symptom score; number of subjects with increased symptom score (21 of 23), and positive reading on discomfor meter (12 of 23), and in 4 individual symptom categories (cough, substantial irritation, wheezing and chest tightness). Three subjects required medication to relieve symptoms. No apparent effects next day or week.	Indicates bronchoconstriction Functional changes, symptoms indicate clinical significance.	Linn et al. (1983b)
0.75 ppm	3h 10 min. exer. at beginning	17	Chamber 22°C, 85% AH = 16.5	Exercising 45 l/min	No clean air control. With SO_2, SR_{aw} increased 263%, FEV_1 decreased 20% after exercise (SR_{aw} increased 322% in second series with no spirometry). Symptom scores increased after exercise. SR_{aw} and symptom scores were not significantly elevated after 1h of recovery in SO_2.	Indicates that recovery is complete for most subjects within 1h of SO_2 + exercise-induced bronchoconstriction.	Hackney et al. (1984)
0.75 ppm	10 min.	23	Chamber 23°C, 90% RH 1) oralnasal 2) mouthpiece AH = 18.5	Exercising V_E = 40 l/min	In clean air, SR_{aw} increased 54% by either oronasal or mouthpiece breathing. In SO_2, SR_{aw} increased 186% oronasal breathing and 321% by mouthpiece. Decline in FVC, FEV_1, PEFR, and $V_{max\ 25-50}$ for both exposure routes. Sig. increase in symptom score, both routes. SR_{aw} increase sig. greater for oral exposures; symptoms and other functional measure changes greater for oral, but not sig. so.	Indicates oronasal breathing ameliorates bronchoconstrictive effects of SO_2, but less effective against symptoms. Functional changes and symptoms indicate clinical significance.	Linn et al. (1983a)

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TABLE 7. SUMMARY OF RESULTS FROM CONTROLLED HUMAN EXPOSURE STUDIES OF PULMONARY FUNCTION EFFECTS ASSOCIATED WITH EXPOSURE OF ASTHMATICS TO SO_2 (continued)

Conc.	Duration	Number of Subjects	Exposure Mode	Exposure Status	Observations	Comments	References
0.75 ppm	10 to 40 min	10	Chamber- 23°C 70% RH AH = 14.4	Exercising V_E = 35 l/min	Significant changes in FEV_1, (mean = -8% MEF_{40} (mean = -22%), V_{max}^{50} ($\bar{x}$ = -11%), and RAW ($\bar{x}$ = 40%). No sig. discomfort persisted 10 min. after exposure. Apparent large increase in lower airway symptom complaints. Wide variable responses among subjects.	Indicates bronchoconstriction Symptoms, functional changes and additive effects of clinical significance began between 0.5 and 0.75 ppm for this study group and conditions on average.	Schacter et al. (1984)
0.75 ppm	10 min.	10	22°C 75% RH AH = 14.6	Mouthpiece Exercising V_E = 34 l/min	Premedication with albuterol blocked the 15% decrease in FEV_1 which occurred with SO_2. Albuterol also caused a 6-8% increase in baseline $FEV_{1.0}$	No symptoms. Albuterol prevented SO_2-induced bronchostriction.	Koenig et al. (1987)
1.0 ppm	30 min. rest 10 min. exercise	9	22°C 75% RH AH = 14.6	Mouthpiece 5-6 x rest V_E (30-50 l/min)	$FEV_{1.0}$ (-23%), V_{max}^{50} (-51%), V_{max}^{75} (-61%), R_T (+71%). Recovery was slower than after 0.5 ppm exposures.	Suggests that more severe SO_2-induced bronchoconstriction requires longer recovery than less pronounced changes at lower conc.	Koenig et al. (1983b)
1.0 ppm	10 to 40 min.	10	Chamber- 23°C 70% RH AH = 14.4	Exercising V_E = 35 l/min	Significant changes in FEV_1 (mean = -14%), MEF_{40} (mean = 27%), V_{max}^{50} ($\bar{x}$ = -22%), and RAW ($\bar{x}$ = -54%). No sig. decrements persist 10 min. after exposure. Apparent large concentration-related increase in lower airway symptom complaints. Three subjects apparently nonresponsive (based on FEV_1) even at this conc., with at least one very sensitive subject showing > 50% FEV_1 decline.	Indicates bronchoconstriction. Symptoms and functional changes suggestive of clinical significance.	Schacter et al. (1984)

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TABLE 7. SUMMARY OF RESULTS FROM CONTROLLED HUMAN EXPOSURE STUDIES OF PULMONARY FUNCTION EFFECTS ASSOCIATED WITH EXPOSURE OF ASTHMATICS TO SO_2 (continued)

Conc.	Duration	Number of Subjects	Exposure Mode	Exposure Status	Observations	Comments	References
1.0 ppm	10 to 75 min.	27	Chamber- 26°C 70% RH AH = 17.1		Sig. decrease in SR_{aw} after all 3 exercise periods but response decreases with time. First Exercise: Increased SR_{aw} in SO_2 (190%) sig. greater than air (47%) Second Exercise: Increased SR_{aw} in SO_2 (147%) sig. greater than air (34%) Third Exercise: Increased SR_{aw} in SO_2 (116%) sig. greater than in air (30%). Group mean symptom analysis for 20 subjects showed sig. increase in shortness of breath and chest discomfort. Substant. variability in subject response; one unable to go beyond 35 min. point.	Respiratory impedence suggests SO_2 induced bronchoconstriction mostly in peripheral airways. Decreased response with time suggest short-term tolerance, but effects of clinical significance occur even after third exercise period.	Roger et al. (1985)
1.0 ppm	1) 10 mins., reported 3 times in succession with 15 minute intervals 2) 30 min. continuous exercise	10	Chamber 26°C 70% RH AH = 17.1	Intermittent Exercise $\dot{V}_E$ = 41 l/min	First Exercise: Significant increase in total SR_{aw} ($\bar{x}$ = 172%). Second Exercise: Sig. increase in total SR_{aw} ($\bar{x}$ = 137%. Third Exercise: Sig. increase in total SR_{aw} ($\bar{x}$ = 106%). Attenuation with time occurred in 4 of 10 subjects. Continuous Exercise: Sig. increase in total SR_{aw} ($\bar{x}$ = 233%) after 30 minutes	Indicates mechanism responsible for apparent tolerance to repeated short-term exposures to SO_2 does not reduce responses to continuous exercise for comparable time periods.	Kehrl et al. (1986)
0.25 to 2 ppm	10 minutes, different days	27	Chamber 26°C 70% RH AH = 17.1	Exercise $\dot{V}_E$ = 42 l/min	Concentration response relationships for four exposures interpolated for each subject to determine $PC(SO_2)$, the SO_2 concentration producing a 100% increase in SR_{aw} over exercise in clean air. Cumulative plot shows 25% of subjects with $PC(SO_2)$ <0.5 ppm, median $PC(SO_2)$ was 0.75 ppm, and about 20% of subjects have a $PC(SO_2)$ of >1.95 ppm.	Reflects additional analyses of data from first exposure period in experiment reported in Roger et al. (1985). Quantifies variability in response among asthmatics for functional changes of potential clinical significance. Suggests effects of concern in some subjects may extend to near 0.25 ppm.	Horstman et al. (1986)

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TABLE 7. SUMMARY OF RESULTS FROM CONTROLLED HUMAN EXPOSURE STUDIES OF PULMONARY FUNCTION EFFECTS ASSOCIATED WITH EXPOSURE OF ASTHMATICS TO SO_2 (continued)

Conc.	Duration	Number of Subjects	Exposure Mode	Exposure Status	Observations	Comments	References
0.125 to 2 ppm	3 minutes doubling exposures in succession with no breaks	8	Oral-mouthpiece 1) -20°C 0% RH 2) 22°C 0% RH 3) 22°C 70% RH 1) AH < 1 2) AH < 1 3) AH = 13.6	Hyperventilation to V_E = 30 to 40 l/min	By design, SR_{aw} increase for clean air alone not sig. Concentration response relationships for 4 to 5 exposures interpolated for each subject to determine PC_{100}, the SO_2 concentration producing a 100% increase over resting baseline. Mean PC_{100} for differing conditions were: Dry Cold Air - 0.51 ppm; Dry Warm, Air - 0.60 ppm; Humid Warm Air = 0.87 ppm; PC_{100} for humid warm air sig. greater than for dry cold or dry warm (which were not sig. diff. from each other.	Nature of doubling concentrations may have affected PC_{100} estimates. Results quantify wide variability among subjects. Indicates very dry air potentiates SO_2 bronchoconstriction regardless of temperature.	Sheppard et al. (1984)
0.25 to 8 ppm	3 min ea. conc.	10	23°C (Dewpoint 15°C) AH = 12.5 RH = 61%	Mouthpiece Isocapnic Hyperpnea $\dot{V}E$ = 40 l/min	Premedication with placebo, 20 mg, or 200 mg cromolyn. SO_2 dose-response to 3 min exposures starting at 0.25. SO_2 dose which increased SRaw by 8 units was 0.35, 0.94, and 1.98 ppm respectively.	Cromolyn decreased airway reactivity to SO_2. High dose of cromolyn caused increased response to methacholine.	Myers et al. (1986a)
0.25 to 8 ppm	3 min ea. conc.	10	23°C (Dewpoint 15°C) AH = 12.5 RH = 61%	Mouthpiece Isocapnic Hyperpnea $\dot{V}E$ = 40 l/min	Premedication with 200 mg cromolyn plus 2 mg atropine was more effective than either drug alone in inhibiting SO_2-induced bronchoconstriction. SO_2 dose which increased SRaw by 8 units was 1.16 (atropine) 1.20 (cromolyn) or 3.66 (both).	Similar effect on dry air hyperpnea-induced bronchoconstriction. Reproducibility of SO_2 dose response was poor.	Myers et al. (1986b)

AH = absolute humidity = g H_2O vapour/m^3 of air.

g/m^3 = mg/ℓ.

HIB = Hyperventilation Induced Bronchoconstriction

In a subsequent paper, Witek et al. (1985) described the symptoms experienced by the subjects in the Schachter et al. (1984) study. Both asthmatics and normal subjects experienced increased respiratory symptoms following SO_2 exposure. Normal subjects complained chiefly of upper airway (nose and mouth) symptoms of odor or unpleasant taste; these symptoms were not increased by exercise. Normals experienced no significant lower respiratory symptoms. There was an increase in lower respiratory symptoms in asthmatics at 0.75 and 1.0 ppm SO_2, although the significance of this trend is not clear ($p = 0.09$). Upper airway symptoms tended to be elevated in both asthmatics and normals, but more consistently in normals. The lower respiratory symptoms increased with exercise in the asthmatics and were significantly correlated ($r = 0.67$, $p < .05$) with the decrease in $FEV_{1.0}$. In contrast, exercise did not affect symptoms in normals. The authors stated that even the asthmatics' symptoms were generally mild and required no therapy.

Linn and coworkers (1983b) also evaluated the responses of naturally breathing asthmatics exposed to SO_2 in an environmental chamber (23°C, 85 percent RH) while performing 5 min of moderately heavy exercise ($\dot{V}_E = 48$ L/min). Twenty-three mild asthmatics (some of whom were hyperreactive to methacholine and all of whom were reactive to 0.75 ppm SO_2) were exposed four times, once each to 0, 0.20, 0.40, and 0.60 ppm. Significant increases in SRaw occurred after clean air exposure due to exercise-induced bronchoconstriction. The SRaw increase after 0.20 ppm was not significantly larger than after clean air, but the SRaw following exposure to the two higher concentrations was significantly elevated. SRaw demonstrated a significant trend to increase with increasing SO_2 concentration but this trend was not linear; the mean increases in SRaw after 0.2, 0.4 and 0.6 ppm SO_2, over those seen with clean air, were 0.54, 2.03, and 6.77 cm $H_2O \cdot$sec. The response data are suggestive of a threshold concentration for response to SO_2. There is a strong possibility of a concentration threshold for SO_2 at low concentrations and ventilations since the scrubbing of SO_2 by the upper airway mucosal surfaces may be so efficient that only a relatively small quantity of SO_2 reaches the reactive portions of the airways.

Roger et al. (1985) studied 27 mild asthmatics (methacholine sensitive, not using cromolyn or steroid medication). Exposures were to 0.0, 0.25, 0.50, and 1.0 ppm SO_2 in an environmental chamber (26°C, 70 percent RH) utilizing natural breathing while performing treadmill exercise ($\dot{V}_E = 41$ L/min). The increases in SRaw post-exercise associated with these exposures were 48, 63, 93, and 191 percent respectively; the increases at the two highest concentrations were significantly greater than with air. The data reported by Roger et al. (1985) were further analyzed (Horstman et al., 1986) in order to determine individual SO_2-SRaw dose-response relationships. This analysis included previously unreported data on exposure to 2 ppm in subjects who were nonresponsive to lower concentrations. From interpolation of the dose-response plots, the concentration of SO_2 which provoked a 100 percent increase in SRaw ($PCSO_2$) was determined for each subject. All SO_2 responses were corrected for the response observed with clean air, i.e., exercise-induced bronchoconstriction. For the most reactive 80 percent of the subjects the $PCSO_2$ ranged from 0.28 to 1.38 ppm; it was greater than 1.95 ppm (and therefore basically indeterminate since the peak exposure level was 2.0 ppm) in the remaining 20 percent of subjects. (This percentage of SO_2-insensitive asthmatics is in general agreement with Linn et al., 1984b) The median $PCSO_2$ in all subjects and under these conditions was 0.75 ppm; 25 percent (i.e., 6) of the subjects had a $PCSO_2$ less than 0.50 ppm, the lowest being 0.28 ppm. The dose-response relationships relate only to the level of exercise used in this study. Different dose-response relationships would be expected for different exercise levels or different exposure durations.

4.3.2 SO_2-Induced Versus Nonspecific Airway Reactivity

It is well established that most asthmatics are highly reactive to bronchial inhalation challenge with histaminergic (histamine) and cholinergic (acetylcholine, carbachol, methacholine) agents. Clear evidence has also emerged that asthmatics are substantially more reactive to SO_2 than are healthy subjects. The relationship between SO_2-induced bronchoconstriction and nonspecific airway reactivity has been examined or alluded to in a number of studies (Horstman et al., 1986; Witek and Schachter, 1985; Sheppard et al., 1983). Airway reactivity to methacholine and to histamine are well correlated ($r = 0.70$) (Chatham et al., 1982). Methacholine reactivity was more highly correlated with exercise-induced bronchoconstriction and was better able to distinguish between normals and asthmatics (Chatham et al., 1982).

Witek and Schachter (1985) reported that the methacholine reactivity of a group of 8 asthmatics was highly ($r = 0.86$, $p < 0.05$) correlated with their reactivity to SO_2. The subjects were a subgroup of 8 of the 10 subjects used in the Schachter et al. (1984) study (see Schachter et al., 1984, for protocol details).

The dose of methacholine required to produce a 20 percent drop in the maximal expiratory flow at 40 percent VC above RV on a partial expiratory maneuver (MEF40%-P) was determined. From the MEF40%-P vs. SO_2 response relationship, the SO_2 concentration required to produce a 20 percent drop was determined. The relationship between the methacholine provocative dose and the SO_2 provocative concentration was determined by rank correlation. This study suggests that there is a relationship between methacholine reactivity and severity of SO_2-induced bronchospasm.

On the other hand, Koenig and Pierson (1985) concluded in their recent review article that the response to a methacholine challenge was not a good predictor of the degree of SO_2-induced bronchoconstriction in asthmatics. They suggested that a positive response to an exercise challenge was more likely to predict a positive response to SO_2. Linn et al. (1983b) present subject data (their Table 1) for methacholine reactivity, exercise response (SRaw change), and SO_2 response (SRaw change), which are sufficient to allow calculation of correlation coefficients between these three variables. The rank-order correlation coefficient between methacholine reactivity and SO_2 response was 0.38, between exercise response and SO_2 response was 0.46, and between exercise and methacholine response was 0.47 (these calculations by the authors of the addendum). The latter two correlation coefficients were significant ($p < 0.05$) and this observation supports the suggestion of Koenig and Pierson (1985). Horstman et al. (1986) have compared the methacholine reactivity (interpolated dose causing a doubling of SRaw) with the SO_2 response ($PCSO_2$; see previous section). The methacholine and SO_2 responses were significantly but weakly correlated ($r = 0.31$).

The relationship of histamine reactivity to SO_2-induced bronchoconstriction is less well described. "Tolerance" to SO_2 exposure reported by Sheppard et al. (1983) was not accompanied by any decrease in histamine reactivity. However, this does not necessarily indicate the absence of an overall relationship between histamine reactivity and SO_2 responsiveness.

One problem in establishing the strength of the relationship between non-specific airway reactivity and SO_2 response is the restricted range of the observations in these studies which deal only with the most reactive segment of the population, namely asthmatics. Inclusion of data from normal subjects would undoubtedly result in a higher correlation. Nevertheless it is apparent that increased SO_2 responsiveness in asthmatics cannot simply be ascribed to elevated non-specific airway reactivity.

4.3.3 Oral, Nasal, and Oronasal Ventilation

For SO_2 in particular, but also for many other gases and aerosols, the inhalation route is an important factor in delivery of the substance to the lung. Since 1982, a number of studies have been reported which specifically address this issue. There are important interactions between the inhalation route, which in many cases is simultaneous oral and nasal breathing (oronasal) (Proctor et al., 1981), and the ventilation rate such that the efficiency of the oral or nasal mucosa in absorbing SO_2 declines as the air flow increases. Approximately 80 percent of the adult population breathes nasally at rest, with some 10 to 20 percent breathing oronasally (Cole, 1982). As noted in the previous addendum (U.S. EPA, 1982c) the studies of Kirkpatrick et al. (1982) and Linn et al. (1982b in the earlier Addendum I; 1983a in the present reference list) indicated the importance of oronasal airway scrubbing of SO_2 in mitigating the effects of SO_2 during nasal or oronasal breathing.

In an effort to further resolve the interaction between exercise ventilation and route of inhalation in asthmatics, Bethel et al. (1983b) studied 9 mild asthmatics breathing humidified air (23°C, 80 percent RH) through either a mouthpiece or a divided facemask (ventilation could be measured separately in nasal and oral chambers). Subjects worked at 250 ($\dot{V}_E$ = 26 L/min), 500 ($\dot{V}_E$ = 53 L/min), or 750 kpm ($\dot{V}_E$, 62 L/min) and breathed either clean air or 0.50 ppm SO_2 for 5 min. Mouthpiece inhalation of SO_2 resulted in increased SRaw at moderate (231 percent) and heavy (306 percent) workloads, but with facemask breathing, the SRaw only increased at the heavy workload (219 percent workload). The oral component of ventilation during mask breathing was estimated to be approximately 38 L/min at the heavy workload, similar to the oral ventilation of 41 L/min with mouthpiece breathing at the moderate workload; the similarity of SRaw responses in these two cases is noteworthy. From these studies it is apparent that oronasal breathing ameliorates some of the effect of SO_2 breathing in asthmatics, but this effect becomes less important as the exercise workload increases and both the overall ventilation rises and the relative contribution of oral ventilation to total ventilation increases.

Kleinman (1984) has modeled the bronchoconstriction response to SO_2 in relation to ventilation, oral/nasal partitioning of ventilation, and differences in SO_2 scrubbing capability of the two upper airways. This model suggests that differences in response to SO_2 can be quantitatively accounted for by differences in penetration of SO_2 to target sites within the lower or thoracic airways (defined as structures at or just below the laryngopharynx).

Because of the possible interference with oral breathing during the facemask exposures, Bethel et al. (1983a) studied 10 mild asthmatics exposed to 0.50 ppm SO_2 in an exposure chamber (23°C, 80 percent RH) to determine if freely breathing subjects would develop bronchoconstriction at this concentration. Following 5 min exercise at 750 kpm ($\dot{V}_E$ unreported, approximately 50 to 60 L/min), SRaw increased 39 percent in clean air but increased 238 percent in 0.50 ppm SO_2 similar to that previously observed with facemask breathing. Thus mild asthmatics performing moderate to heavy exercise exhibited clear evidence of bronchoconstriction after 5 min exposure to 0.50 ppm SO_2 while breathing unencumbered.

In a subsequent study (Bethel et al., 1985), the effects of 0.25 ppm SO_2 were studied in 19 mild to moderate asthmatics using a similar protocol (23°C, 36 percent RH with 5 min exercise at 750 kpm). SRaw increased from 6.4 to 11.3 post-exercise in clean air and from 5.7 to 13.3 post-exercise in 0.25 ppm SO_2. The slightly greater response following SO_2 exposure was apparently significant ($p < 0.05$, Wilcoxon one-tailed sign test). The application of a signed rank test, preferable in this case, would not confirm this significance. However, when the workload was increased to 1000 kpm in 9 of the 19 subjects, the increase in SRaw after clean air exercise was slightly, but not significantly, greater than that after exercise with 0.25 ppm SO_2. The authors suggested that the threshold concentration of SO_2 which may cause bronchoconstriction in mild asthmatics under conditions of moderate to heavy exercise appears close to 0.25 ppm. However, the very small rise in SRaw at only one work output indicates that the additional effect of 0.25 ppm SO_2 (over that produced by exercise) is of minor, if any, clinical significance. Nevertheless, it must be stressed that these asthmatics had relatively mild disease.

Koenig et al. (1983b) examined the effects of exposure to 0.5 and 1.0 ppm SO_2 combined with a sodium chloride droplet aerosol in nine extrinsic adolescent asthmatics. Judging from their medication requirements, this group of asthmatics would have to be considered more severe than the adult asthmatics studied by several other investigators. The exposures were delivered via mouthpiece (22°C, 75+ percent RH) for 10 min during moderate treadmill exercise (30 min rest exposures were followed by 10 min exercise). The responses ranged from a 15 percent decrease in $FEV_{1.0}$ at 0.5 ppm to a 61 percent decrease in $\dot{V}_{max75}$ at 1.0 ppm. The response to 1.0 ppm tended to be greater but this difference between SO_2 concentrations did not attain overall statistical significance. Nevertheless, the effects of SO_2 on lung function persisted longer after the higher concentration exposure. $FEV_{1.0}$, $\dot{V}_{max50}$ and $\dot{V}_{max75}$ (partial flow volume curves) were significantly reduced and total respiratory resistance (forced oscillation) was significantly increased following mouthpiece breathing of 0.5 or 1.0 ppm SO_2. Seven of nine subjects were also exposed to 0.5 ppm SO_2 plus aerosol delivered via a facemask (ventilation 5 to 6 times rest or 30 to 50 L/min). The pulmonary function changes after breathing 0.50 ppm SO_2 plus aerosol via facemask were not significantly different from baseline. However, some of the subjects intentionally breathed through their nose rather than oronasally; therefore the comparison of the results of this study with those of Bethel et al. (1983a) would not be appropriate.

Previous studies (Andersen et al., 1974) cited in the criteria document have suggested that nasal resistance increases following SO_2 exposure. Because this could have an important impact on the route of inhalation and/or the oronasal ventilation switch point, Koenig and associates (1985) examined the effects of 0.50 ppm SO_2 on the work of nasal breathing in a group of moderate adolescent asthmatics (7/10 were theophylline users). Subjects were exposed to SO_2 (and H_2SO_4 aerosol -- 100 $\mu g/m^3$) either via mouthpiece or oronasal facemask (22°C, 75 percent RH). Thirty min resting exposure was followed by 20 min of moderate exercise on a treadmill ($\dot{V}_E$ = 43 L/min). Exposure to SO_2 via mouthpiece or facemask resulted in an approximate 30 percent increase in nasal work of breathing (measured with a divided diving mask containing two pressure transducers which measured the pressure drop across the nasal passages). Due to marked inter- and intra-individual variability in these nasal measurements, only the increase in nasal work of breathing after facemask exposure, measured at 22 min post-exercise, was found to be statistically significant. Exercise, per se, is associated with a reduction in nasal resistance which persists for about 5 to 15 minutes after exposure ceases (Forsyth et al., 1983). The effects of SO_2 on nasal resistance may therefore be offset by the effects associated with exercise leading perhaps to minimal changes in nasal resistance immediately post-exercise. No changes in nasal resistance were observed after clean air or sulfuric acid aerosol exposure. The decreases in $FEV_{1.0}$ and $\dot{V}_{max50}$ were significantly greater with mouthpiece than with facemask exposure to 0.50 ppm SO_2. The implications of this finding are not clear at present. If SO_2 raised nasal resistance during exercise (and this is not presently known), the relative proportion of oral ventilation could increase. However, the reduction in nasal resistance associated with exercise may override or negate the effect of SO_2 on nasal resistance. Increased oral breathing of SO_2 could result in a greater delivery of SO_2 to airways below the larynx.

4.3.4 Time Course of Response to SO_2 in Asthmatics

Early studies of SO_2 exposure in normal healthy subjects indicated that the peak response occurred early in exposure and was reduced with continued exposure. The effect of prolonged or repeated exposure has recently been addressed in asthmatics.

Sheppard and associates (1983) reported the responses of mild to moderate asthmatics (n = 8) exposed three consecutive times to 0.5 ppm SO_2. The subjects performed voluntary eucapnic hyperpnea with 0.5 ppm SO_2 for 3 min at a ventilation which had previously caused bronchoconstriction (air temperature = 22.6°C, RH = 82 percent). Three subjects failed to reach the target of a 60 percent increase in SRaw above baseline and consequently performed additional hyperpnea to produce increased SRaw. Twice more, at 30-min intervals, the SO_2 hyperpnea was repeated. SRaw was measured before and after each SO_2 exposure. A single bout of SO_2 hyperpnea was performed on the following day and again one week later. The first exposure to SO_2 caused a doubling of SRaw (104 percent increase). The second and third SO_2 exposures elicited only modest increases in SRaw (35 percent, 30 percent respectively). However, 1 day and 7 days later, the response to SO_2 was similar (+89, +129 percent) to that on the first exposure.

In this study, the relationship of SO_2 tolerance to histamine-induced bronchoconstriction was examined in a subgroup of four subjects. A baseline histamine challenge test was followed 30 min later by two 3-min periods of SO_2 breathing separated by 30 min (as in the initial part of the study). When the histamine challenge was repeated after a further 30 min, the histamine dose-response relationship was unchanged despite the blunted response to SO_2 inhalation. This study demonstrated that repeated exposure of asthmatics to 0.5 ppm SO_2 by mouthpiece at 30-min intervals resulted in a blunted SO_2 response (tolerance) which persisted for at least 30 min but was absent after 24 h and was not associated with any change in airway reactivity to histamine. The implications of this study for response mechanisms are discussed in Section 4.4.

Linn et al. (1984c) also studied the effect of repeated SO_2 inhalation in 14 mild to moderate asthmatics who were exposed to 0.6 ppm SO_2 for 6 h on each of two consecutive days. These were compared with similar clean air exposures. They performed two 5-min bouts of exercise ($\dot{V}_E$ = 50 L/min), one immediately upon entering the exposure chamber (22°C and 85 percent RH) and a second bout 5 h later. SRaw was measured immediately post-exercise and at hourly intervals between exercise periods. With SO_2 exposure, SRaw was approximately doubled following each exercise bout. Small increases in SRaw also occurred following exercise in clean air. There were no differences in response between early and late exercise challenges and no significant differences in SRaw response between exposure days. SGaw, but not SRaw, responses indicated smaller decreases on the second SO_2 exposure day ($-0.091\ sec^{-1} \cdot cmH_2O^{-1}$) than the first ($-0.119\ sec^{-1} \cdot cmH_2O^{-1}$). This difference was of only marginal statistical significance and not of any clinical importance. The results of this study indicate that SO_2-exercise challenges separated by 5 h (between exercise periods) produce essentially similar responses and that the responses are not appreciably different on two consecutive days. The Linn et al. (1984c) and Sheppard et al. (1983) studies had several methodological differences; respectively, these were free breathing vs. mouthpiece, exercise vs. eucapnic hyperpnea, 4.5 h vs. 30 min interexposure interval, 5 min vs. 3 min exposure duration, and 0.6 ppm vs. 0.5 ppm SO_2 concentration. Nevertheless, in each study, an initial SO_2 exposure which produced at least a doubling of SRaw was followed later by a second exposure. With the shorter 30-min interval in the Sheppard study, the response to SO_2 was blunted. However, with the longer 5-h interval in the Linn study, the SO_2 response was unchanged from the initial exposure. Evidence from the exercise-induced bronchoconstriction literature (Edmunds et al., 1978; Stearns et al., 1981) indicates that the refractory period following exercise induced bronchoconstriction persists for 2 to 4 h. The refractory period following SO_2-induced bronchoconstriction lasts at least 30 min but less than 5 h.

Snashall and Baldwin (1982) studied the effect of exposures to 8 ppm SO_2 repeated at 4 h and 24 h in 4 normal and 1 asthmatic subjects. Compared to the initial exposures, SO_2-induced bronchoconstriction was reduced 42 percent at 4 h while no difference was observed at 24 h.

In a more comprehensive examination of repeated exercise during continuous SO_2 exposure in a large subject population (n=28) exposed to 3 different SO_2

levels with repeated exercise, Roger et al. (1985) also observed attenuation of SO_2-induced bronchoconstriction. The subjects worked at a moderate workload ($\dot{V}_E$ = 42 L/min) and breathed freely (except for 2 min at the end of exercise periods 2 and 3). They were not selected for SO_2 sensitivity, were sensitive to methacholine challenge, and used no cromolyn or steroids. Each subject was exposed, on three different days, to three SO_2 concentrations (0.25, 0.50, and 1.0). During each exposure, the subject exercised three times for 10 min each separated by 15-min intervals between exercise bouts. SRaw was measured pre-exposure and following each exercise period. After the first exercise, SRaw increased significantly over that seen with clean air (48 percent), with exposure to both 0.5 (+93 percent) and 1.0 ppm SO_2 (+191 percent). With subsequent exercise bouts in both 0.5 and 1.0 ppm SO_2, the SRaw increased only about half as much (third exercise SRaw increase was 52 percent and 116 percent in 0.5 and 1.0 ppm, respectively). This attenuation of response was less than that seen by Sheppard et al. (1983). Nevertheless, there were several differences between the two studies (exposure duration 3 min vs. 10 min, inter-exposure interval 30 min vs. 15 min, mouthpiece eucapnic hyperpnea vs. free breathing exercise, SO_2-sensitive vs. methacholine-sensitive selection criterion). The Roger et al. (1985) subjects demonstrated a refractoriness to both exercise in clean air and to exercise in SO_2; the latter was of greater absolute magnitude in terms of less increase in SRaw but the relative reduction in response from first to last exercise periods was similar for repeated exercise in either clean air or SO_2.

A subset of 10 subjects from the Roger et al. (1985) study were further studied by Kehrl and coworkers (1986). The subjects were selected for moderate SO_2 sensitivity (i.e., no subjects nonresponsive to SO_2 were used and the most reactive subjects were not studied). In addition to the three 10-min exercise periods performed previously, these subjects exercised continuously for 30 min at the same exercise intensity ($\dot{V}_E$ = 41 L/min) in an environmental chamber (26°C, 70 percent RH) while exposed to 1.0 ppm SO_2. The SRaw data for the original intermittent exercise exposures were similar to those of the original larger subject group (SRaw: baseline, 5.4; post-exercise-1, 14.7; post-exercise-2, 12.8; post-exercise-3, 11.1). After 30 min continuous exercise in 1.0 ppm SO_2, SRaw significantly increased from 5.2 to 17.3 cm H_2O·sec. The SRaw change was not significantly different than that seen after the first 10 minute exercise period of the intermittent exercise exposure. This study demonstrated that SO_2-induced bronchoconstriction is elicited by 10-min exposures but a further 20 min of continuous exercise resulted in only a slightly greater increase in SRaw which did not attain statistical significance.

In order to examine the time course of recovery from SO_2-induced bronchoconstriction in asthmatics, Hackney et al. (1984) exposed 17 mild to moderate, nonsmoking, SO_2-sensitive asthmatics (not using cromolyn or steroid medication) to 0.75 ppm SO_2 for 3 h. A secondary objective of this study was to determine the usefulness of spirometric testing as an adjunct or alternative to plethysmography under such exposure conditions. The exposure consisted of 3 h in an environmental chamber with a 10-min exercise period ($\dot{V}_E$ = 45 L/min) at the beginning of the exposure followed by post-exercise and hourly SRaw measurements. SRaw was approximately quadrupled (+263 percent) after exercise, returned almost to baseline at one hour (+34 percent, not significant) and was unquestionably back to baseline after 2 h recovery. In an otherwise identical exposure sequence which included spirometric testing, the $FEV_{1.0}$ was significantly reduced (-20 percent) post-exercise. The correlation between the $FEV_{1.0}$ and SRaw changes was significant ($r = 0.60$) but accounted for considerably less than half the variance, indicating that the two measures did not track each other closely in all subjects. This study demonstrated that moderate SO_2/exercise-induced bronchoconstriction will be relieved during rest (over a 1 to 2 h period) even if a low-level SO_2 exposure is continued. Second the authors demonstrated that changes in $FEV_{1.0}$ are also useful indicators of SO_2 exposure in asthmatics, although it is not clear that significant changes in $FEV_{1.0}$ would occur with less severe exposure more typical of the ambient environment.

4.3.5 Exacerbation of the Responses of Asthmatics to SO_2 by Cold/Dry Air

It has been well established that both cold air and dry air can exacerbate bronchoconstriction in asthmatics (Deal et al., 1979a; Strauss et al., 1977). The precise mechanism(s) for the effect are not universally agreed upon (Anderson, 1985). Although direct convective cooling of the airway plays a minor role, the major avenue of heat loss is due to evaporation to humidify the inspired air. Evaporation of airway surface liquid may also lead to other changes discussed in section 4.4. The potential for evaporative cooling by inhaled air can be most readily appreciated from the determination of the absolute humidity of the inspired air. Absolute humidity (AH) expresses the water content of the air in mg/L (g/m^3). The lower the AH, the greater the potential for evaporative cooling. AH is listed, for each study, in Table 4. For reference, the AH of saturated air at 37°C (i.e., BTPS) is 44 mg/L. Therefore, in order to bring inspired air at 0°C, AH = 1 mg/L, to BTPS, the temperature of each liter of air must be increased to 37°C (0.011 kcal) and 43 mg of water must be evaporated (0.025 kcal) (calculated from the respiratory heat exchange equation of Deal et al., 1979b).

Sulfur dioxide exposure can occur during the winter months when the ambient air temperature is low, and consequently the water vapor content is reduced. Accordingly, Bethel and coworkers (1984) examined the separate and combined effects of sulfur dioxide and cold dry air in seven asthmatics (mild to moderate asthma) breathing via mouthpiece. In this study and the following study by Sheppard and coworkers (1984), a series of bronchoprovocation tests were used. The methods are as follows:

> The subjects breathed a test gas mixture for 3 min, then SRaw was determined every 30 s for 2 min. This cycle of 3 min exposure and 2 min SRaw testing was repeated until the desired response was achieved. The ventilatory bronchoprovocation test consisted of performing voluntary eucapnic hyperventilation at increasing ventilation levels (20, 30, 40, 50, 60, etc. L/min) while breathing a single test gas mixture. The SO_2 bronchoprovocation test consisted of breathing (eucapnic hyperventilation) at some fixed ventilation and gas temperature and humidity with successively doubling levels of sulfur dioxide (e.g., 0, 0.125, 0.25, 0.50, 1.0, 2.0 ppm SO_2) used as the stimulus.

Bethel's subjects performed ventilatory bronchoprovocation tests with both 0.50 ppm SO_2 in warm humid air and with no SO_2 in cold-dry air (-11°C, dew point -15°C) until an increase in SRaw was observed in order to determine the ventilation which caused "little or no bronchoconstriction" with either stimulus. At the selected ventilation, subjects breathed on a mouthpiece for 3 min one of the following mixtures: (1) warm-humid (23°C, dew point = 18.4°C) air, (2) warm humid air with 0.50 ppm SO_2, (3)cold dry air, (4) cold dry air with 0.50 ppm SO_2. Modest but nonsignificant increases in SRaw followed each of the first three conditions [(1) +3 percent, (2) +38 percent, (3) +18 percent]. However, the combination of 0.50 ppm SO_2 and cold dry air caused a striking increase in SRaw (from 6.94 to 22.35, or a 222 percent increase). In this study, the combined effect of breathing cold dry air and 0.50 ppm SO_2 via mouthpiece was clearly larger than the sum of the individual response to either SO_2 or cold dry air.

Sheppard and coworkers (1984) further explored the interaction of breathing cold dry air and SO_2 via mouthpiece in a group of 8 mild asthmatics. The purpose of the study was to determine the relative contributions of decreased air temperature (-20°C) and reduced water vapor content (0 percent RH). Using a ventilatory bronchoprovocation test with cold dry air, the highest ventilation which did not cause increased SRaw was determined. The study consisted of having the subjects perform eucapnic voluntary hyperpnea, at the selected ventilation, 6 consecutive times for 3 min at a time with 2 min intervals between efforts. This was done on four separate occasions (different days) ordered randomly. On one occasion, the subject breathed cold-dry air only; this did not cause an increase in SRaw. The three other tests consisted of SO_2 bronchoprovocation tests at the selected ventilation with successive doubling of SO_2 concentrations (starting at 0.125 ppm), one with cold-dry air, one with warm-dry (22°C, 0 percent RH) air, and one with warm-humid (22°C, 70 percent RH) air. The SO_2 concentration required to produce a doubling of baseline SRaw (PC100) was interpolated from the dose-response curve. The PC100 for cold-dry air (0.51 ppm) and for warm-dry air (0.60 ppm) were not significantly different but both were less than the PC100 for warm-humid air (0.87 ppm). The PC100 measured in this study may not be a useful effects index because the response may be a function of the cumulative effect of all SO_2 concentrations breathed, as noted by the authors. In addition, the authors considered the possible mitigating effect of repeated exposure - tolerance, but the importance of this is unclear. Further studies were then performed using a ventilatory bronchoprovocation test while breathing either 0.0, 0.1, or 0.25 ppm SO_2 in warm-dry air. From the ventilation-SRaw dose-response plots at each SO_2 concentration, the ventilation producing an 80 percent increase in SRaw (PV80) was determined. The PV80 at 0.0, 0.1, and 0.25 ppm SO_2 were 54.9, 51.1, and 49.3 L/min, respectively. The differences in PV80 between 0.1 or 0.25 and clean air (0.0 ppm) reportedly reached significance although it was not clear how these data were analyzed (presumably repeated measures analysis of variance). Regardless of whether or not the difference in PV80 between clean air and 0.1 and 0.25 ppm SO_2 was statistically significant, the magnitude of this difference is small and of no established or obvious clinical importance. Nevertheless, the first part of this study did confirm that breathing dry air and cold air potentiates sulfur dioxide-induced bronchoconstriction. This potentiation could be an additive effect since both cooling (convective and evaporative) and drying of the airway may act as direct bronchoconstrictive stimuli, per se (Sheppard et al., 1984). In addition, the drying of the upper airway also reduces the ability of the oropharynx to scrub SO_2 from the inhaled air and may also cause a concentrating effect of the remaining airway surface liquid (see Mechanism section).

Concurrent studies by Linn and coworkers (1984a) also were directed at the possible interaction of inhalation of sulfur dioxide and cold air. They studied a group of 24 mild to moderate SO_2-sensitive asthmatics. A preliminary study to determine the effects of humidity at cold ambient temperatures included eight subjects exposed to 0.0, 0.2, 0.4, and 0.6 ppm SO_2 at 5°C under two humidity conditions (81 percent and 54 percent). The subjects exercised for 5 min in an environmental chamber at a workload selected to elicit a ventilation of approximately 50 L/min (range 37 to 60) and breathed naturally. SRaw showed a tendency to increase more from pre- to post-exposure with increased SO_2 concentration. The post-hoc analyses for changes at each concentration were not presented, presumably because of the small sample size and the non-randomized experimental design. No effect of ambient humidity on response to SO_2 was seen at the 5°C air temperature. However, the difference in water vapor content at the low and high humidities was approximately 1.84 mg/L, approximately 1/20 of the difference in water vapor pressure between ambient and BTPS, and thus the absence of a difference should have been expected. A second study in this same series compared responses of 24 asthmatic subjects exposed to 0.6 ppm SO_2 under warm-humid (22°C, 85 percent RH, AH = 16.5) and cold humid (5°C, 85 percent RH, AH = 3.4) conditions. The same exercise and natural breathing procedures as above were followed. Breathing 0.0 ppm SO_2, subjects had small nonsignificant increases in SRaw under warm (27 percent) and cold (38 percent) conditions. Exposure to 0.6 ppm SO_2 under these temperature-humidity conditions produced significant increases in SRaw in both warm (132 percent) and cold (182 percent) conditions. However, the temperature effect, unlike in the Sheppard et al. (1984) and Bethel et al. (1984) studies, was not significant although the trend was in the direction of an increased response at the lower temperature. The temperature difference between cold and warm air was larger in the Sheppard et al. and Bethel et al. studies (42°C and 34°C, respectively) compared to the Linn et al. study (17°C). However the cold-warm difference in inspired air water content (AH) were similar for the three studies (14.8, 12.6, 13.1 respectively). Nevertheless, it is apparent that the exacerbation of SO_2-induced bronchoconstriction by cold air, containing small quantities of water vapor, is minimal in freely breathing asthmatics exposed during moderately heavy exercise at 5°C air temperature.

In order to determine the possible effects of even colder ambient air temperatures, Linn et al. (1984b) exposed 24 mild SO_2-sensitive asthmatics (including 11 subjects from Linn et al., 1984a) to 0.0, 0.3, and 0.6 ppm SO_2 at +21, +7, and -6°C (RH approximately 78 percent). The exposure duration was 5 min. The authors noted that "only 10-20 percent of clinically asthmatic prospective subjects had to be rejected as non-responsive to SO_2" (10 min exercise at 40 L/min breathing 0.75 ppm SO_2). There was a significant effect of decreasing air temperature and of increasing SO_2 concentration on the post-exercise SRaw. However, the authors reported that there was no statistically significant interaction of air temperature and SO_2 concentration for SRaw although the interaction was apparently significant for SGaw. The effect of cold air (in increasing SRaw or decreasing SGaw) was most pronounced with the 0.0 ppm SO_2 exposures and minimal with 0.6 ppm exposures. The results of this study do not support the hypothesis that SO_2 acts synergistically with cold air in freely breathing, exercising, mild to moderate asthmatics. The authors concluded that the cold air and SO_2 effects "acted additively at most." The results for the 7°C and 21°C 0.6 ppm SO_2 exposures (+207 percent, +150 percent SRaw) were similar to those seen in their previous study (Linn et al., 1984a) (+182 percent, +132 percent SRaw), thus demonstrating the reproducibility of these studies.

In order to study the full range of SO_2-temperature-humidity interactions, Linn et al. (1985a) also examined the effects of warm-dry (38°C, 20 percent RH) and warm-humid (38°C, 85 percent RH) conditions on 22 SO_2-exposed (0.6 ppm) asthmatics. The exposure protocol was similar to the two 1984 studies with a 5 min chamber exercise period and ventilation of approximately 50 L/min. The experimental design was a three-factor (SO_2-0.0 and 0.6 ppm; temperature-21 and 38°C; and humidity-20 percent and 80 percent) factorial design with repeated measures across all factors. In this study, the major differences would be anticipated to occur between the warm humid (38°C, 85 percent RH) condition and the cooler dryer condition (21°C, 20 percent RH). There were significant effects of temperature, SO_2 and humidity on the delta-SRaw (pre- to post-exercise) response and significant temperature-SO_2 and humidity-SO_2 interactions. The largest clean air increase in SRaw (20 percent) occurred with cool-dry air and the smallest with warm-humid. The largest SO_2 induced increase in SRaw (204 percent) occurred under cool-dry conditions and again the smallest change (35 percent) occurred under warm-humid conditions. Symptoms showed a similar pattern of response after SO_2 exposures with lower symptoms scores under warm-humid than cool-dry conditions. SRaw responses to 0.6 ppm SO_2 under 21°C-humid conditions were similar for all three Linn et al. studies (1984a, 132 percent; 1984b, 150 percent; 1985a, 157 percent). The response under warm humid conditions was considerably less. The authors discussed the possibility that they observed a synergism between SO_2 exposure and airway drying/cooling due to reduced temperature or humidity of inspired air.

4.3.6 Clinical Relevance

As discussed in the introduction, there is no obvious or clearcut point where SO_2 effects cease to be a mere annoyance and became an adverse health effect. The "clinical" importance of the various observations cannot be interpreted in an unequivocal fashion. There were no reports of cases where subjects required emergency treatment or hospitalization following SO_2 exposure. Furthermore, there was no evidence reported which indicated that brief SO_2 exposure caused either acute or chronic changes in nonspecific airway reactivity and the majority of subjects recovered spontaneously within an hour. The responses (SRaw and $FEV_{1.0}$) were no greater than those observed with exposure to aeroallergens and no delayed effects of SO_2 were reported.

However, in addition to changes in spirometry, airway resistance, and symptoms of wheezing and chest discomfort; several "clinically" relevant observations were documented. These observations are summarized in Table 8. The "clinical" significance of these responses included the use of medication following SO_2 exposure, the modification of activity, or the inability to complete the SO_2 exposures. The repeatability of such responses is demonstrated, by one subject from the Koenig et al. (1985) study, who participated in two exposure to 0.50 pmm SO_2 and, in both cases, was unable to complete the exposure and required a brochodilator to reverse the bronchoconstriction.

TABLE 8. CLINICALLY SIGNIFICANT RESPONSES

Study Reference	Responses
Bethel et al. (1984)	2/7 subjects required bronchodilator after cold air + 0.50 ppm SO_2
Koenig et al. (1985)	2/10 subjects exposed to 0.50 ppm SO_2 via mouthpiece could not complete exposure; required bronchodilator to reverse bronchoconstriction. One tried again with the same result.
Linn et al. (1984b)	1/24 took isoproterenol after 0.4 ppm SO_2 3/24 took isoproterenol after 0.6 ppm SO_2
Linn et al. (1984a)	1/24 took isoproterenol after 0.6 ppm SO_2
Linn et al. (1984c)	One subject required reduced exercise level to complete exposure at 0.6 ppm SO_2
Roger et al. (1985) (Horstman et al., 1986)	2/28 subjects unable to complete exposure regimen. One dropped out at 0.5 ppm SO_2 (he required medication - anecdotal report) Second subject unable to complete exposure at 1.0 ppm SO_2

4.4 MECHANISM(S)

4.4.1 Mode of Action

A single unequivocal definition of asthma is not realistic on the basis of existing knowledge and the heterogeneity of the disease. The single condition that is common to all definitions of asthma is the reversibility of slowed forced expiration presumably due to airway narrowing (smooth muscle contraction, excess mucous secretion, mucosal edema). Most current definitions of asthma also include the concept of nonspecific airway hyperreactivity (e.g., methacholine, histamine). The present American Thoracic Society definition of asthma is:

> A disease characterized by an increased responsiveness of the airways to various stimuli and manifested by slowing of forced expiration which changes in severity either spontaneously or with treatment.

It is noteworthy that the data summarized in this addendum indicate that asthmatics experience substantial, but transient, bronchoconstriction (slowed forced expiration) when exposed to low SO_2 concentrations (i.e. increased responsiveness).

Because of its relatively rapid reversibility, SO_2-induced bronchoconstriction in asthmatics is likely the result of decreased airway caliber caused by contraction of airway smooth muscle. The study of Roger et al. (1985) indicated the largest SO_2-induced increases in airway resistance measured by plethysmography were associated with increases in the low frequency component of respiratory system impedance measured by the forced random oscillation (noise) technique. The interpretation of this finding was an elevated peripheral resistance associated with constriction of anatomically peripheral or small airways. However, narrowing of central upper airway structures such as the larynx and glottis may accompany increased airway resistance (Cole, 1982) and it is possible that some of the increase in airway resistance may be due to elevated laryngeal or glottal resistance.

Contraction of airway smooth muscle in response to environmental stimuli can be evoked by intrinsic chemical and/or physical stimuli acting via neural and/or humoral pathways. SO_2 may either act directly on smooth muscle or may cause the release of chemical mediators from tissue, especially the release of histamine from mast cells. It is beyond the scope of this document to provide even a brief review of the mechanism of action of all the possible pharmacologic mediators of SO_2-induced bronchoconstriction. However, some plausible candidates include histamine, slow-reacting substance of anaphylaxis, leukotrienes, and prostaglandin F_2-alpha, all of which are released in the airways and can cause smooth muscle contraction.

As reported in the previous addendum (U.S. EPA, 1982c), both activation of parasympathetically mediated reflexes (Nadel et al., 1965; Sheppard et al., 1980) and mast cell degranulation (Sheppard et al., 1981) with consequent release of chemical mediator (most likely histamine) play a significant role in SO_2-induced bronchoconstriction. While the specific mechanism whereby SO_2 interacts with the airways to induce bronchoconstriction has not been elucidated, additional studies relevant to the mechanism(s) have appeared since the previous addendum. These studies assessed the inhibitory effects on SO_2-induced bronchoconstriction of a variety of receptor antagonists (drugs that bind the receptors but do not stimulate the receptor-induced response). Results from these studies suggest that mechanisms in addition to reflex bronchoconstriction and mast cell degranulation may play a significant part in the responses of the asthmatic airway to SO_2.

Snashall and Baldwin (1982) studied the effects of atropine and cromolyn on relatively mild bronchoconstriction (Raw increased <100 percent above baseline) induced by breathing 8 ppm SO_2 at rest. Both atropine and cromolyn at least partially blocked SO_2-induced bronchoconstriction in all but one of 11 normal subjects. The degree of atropine blockade was inversely related to the magnitude of the SO_2-induced response ($r = -0.75$), i.e., small responses were completely blocked, while there was little blockade of large responses. For asthmatics, atropine enhanced SO_2-induced bronchoconstriction in three of four subjects tested; minimal blockade was observed in the remaining subject. Cromolyn blocked the SO_2-induced response in three of the four asthmatic subjects.

Tan et al. (1982) exposed resting normal and atopic subjects to 20 ppm and asthmatics to 10 ppm SO_2 to induce bronchoconstriction. Both ipratropium bromide (IB, an anticholinergic agent similar to atropine) and cromolyn partially inhibited the SO_2-induced response in all normal and atopic subjects tested. For asthmatics, IB had little effect on SO_2-induced bronchoconstriction in five of nine subjects and afforded only partial blockade in the remaining four subjects. Cromolyn at least partially inhibited SO_2-induced bronchoconstriction in all 18 asthmatics tested. Clemastine (a selective H_1 receptor antagonist without anticholinergic or antiserotinergic activity) effectively blocked the SO_2-induced response in five of seven asthmatic subjects tested.

Koenig et al. (1987) studied a group of adolescents with a history of allergy but without clinically documented extrinsic asthma by the authors' criteria. All had exercise-induced bronchospasm, defined as a 15 percent or more reduction in $FEV_{1.0}$ after 6 min exercise at 85 percent of $\dot{V}O_{2max}$. The aim of this study was to determine whether the beta-2 sympathomimetic drug albuterol could inhibit SO_2-induced bronchoconstriction. Following baseline lung function tests, the subjects received either placebo or 180 µg of albuterol aerosol, again performed lung function tests, and then exercised for 10 min ($\dot{V}_E$ = 34 ℓ/min). The subjects were exposed to clean air or 0.75 ppm SO_2 after either placebo or albuterol pretreatment. Pretreatment with albuterol resulted in a 6 to 8 percent increase in $FEV_{1.0}$ and a 17 to 23 percent decrease in resistance (forced oscillation method). In clean air, exercise plus placebo resulted in a 4 percent decrease in $FEV_{1.0}$. In SO_2, exercise plus placebo resulted in a 15 percent decrease in $FEV_{1.0}$ below the preplacebo baseline. Following pretreatment with albuterol, change in $FEV_{1.0}$ did not differ between

SO_2 and clean air exposure and in neither case did $FEV_{1.0}$ drop below the prealbuterol baseline. This study illustrates that SO_2 induced bronchoconstriction can be prevented (or at least the response curve can be shifted) by pretreatment with a beta-2-sympathomimetic drug. Albuterol could act by inhibiting smooth muscle contraction or by inhibiting mast cell degranulation or by a combination of the two effects.

Myers et al. (1986a) examined the effect of cromolyn on SO_2-induced bronchoconstriction in a group of 10 asthmatics. They demonstrated a dose-dependent inhibition of SO_2-induced bronchoconstriction by cromolyn given prior to the SO_2 exposure. It was also established that the cromolyn did not reduce nonspecific airway reactivity. In fact, the methacholine reactivity increased with the higher dose (200 mg) of cromolyn. The mechanism by which cromolyn exerts this inhibitory effect on SO_2-induced bronchoconstriction is not established but could result from the inhibitory effect of cromolyn on mast cell degranulation.

In a subsequent study (Myers et al., 1986b); the effects of chromolyn plus atropine or SO_2-induced bronchoconstriction were studied in 9 subjects, 7 of whom participated in the previous study. It was demonstrated that the combination of atropine and chromolyn was more effective at inhibiting SO_2-induced bronchoconstriction than either agent above. The SO_2 dose-response curve was not reproducible for some subjects who participated in both studies.

4.4.2 Breathing Mode and Interaction With Dry Air

There is no question that the magnitude of SO_2-induced bronchoconstriction is significantly greater with oral than with oronasal or nasal breathing (Kirkpatrick et al., 1982). When SO_2 is inhaled by mouth more SO_2 penetrates beyond the pharynx to sites involved in the induction of bronchoconstriction (Bethel et al., 1983b; Kleinman, 1984). It is assumed that because of their geometry and greater relative surface area, the nasal passages are capable of effectively removing most SO_2 breathed at rest and a large percentage during conditions of increased ventilation (exercise, isocapnic hyperpnea). While there is certainly less relative surface area available for SO_2 scrubbing in the oral cavity, other factors may also influence increased bronchoconstriction associated with mouth breathing of SO_2, especially at higher ventilation rates.

Increased oral ventilation may result in substantial drying of both upper (oral and pharyngeal area) and lower (larynx and trachea) airways. The extent of airway surface drying will depend upon the ventilation (air flow rate) and water content of inhaled air. Airway drying could lead to alterations in both the quantity and properties of surface liquid in the airways. It is not known whether changes in the volume or water content of the surface liquid lining the upper airway will result in altered SO_2 uptake or the penetration of the gas to sites in the intrathoracic airway more likely involved in the induction of bronchoconstriction. Another factor which is altered by drying of airway surface liquid is its osmolarity. Hyperosmolar solutions can induce bronchoconstriction (Anderson, 1985) but it is not known whether changes in mucous osmolarity may affect the functional response to SO_2.

Two laboratories (Cardiovascular Research Institute, UCSF, and Rancho Los Amigos Hospital) have performed the bulk of the work on the interaction of SO_2 breathing and inhaled air temperature and humidity. Although the results of the two labs have been qualitatively similar, the mouthpiece breathing studies (e.g., Bethel et al., 1983b) have typically yielded more pronounced increases in airway resistance. In SO_2 exposures using oronasal ventilation, interlaboratory differences have been smaller. The use of mouthpiece breathing results in a more direct airflow path of lower resistance than does unencumbered oronasal breathing (Proctor et al., 1981; Cole, 1982). Under situations of unencumbered oronasal breathing, the mouth may act as an effective organ of air modification (i.e., warming, humidifying, scrubbing of particles and soluble gases). During mouthpiece breathing, this effectiveness is reduced because of the alteration in oral airway geometry and the bypassing of some of the oropharyngeal surfaces involved in air modification. Thus some of the difference between laboratories may be due to differences in the amount of airway drying and the volume of nasal ventilation, both of which would favor greater upper airway SO_2 scrubbing in studies using oronasal ventilation. Undoubtedly subject selection criteria and medication also play an important role in the magnitude of response but such differences between study series are not obvious (see subject table). Another possibility, noted incidentally by Koenig et al. (1985), is that subjects may deliberately breathe via the nasal airway, despite the higher resistance, in order to alleviate both the drying effect due to cold (and/or dry) air and the effect of SO_2 which may be associated with the distinctive odor or taste.

Cole (1982) notes that approximately 85 percent of adults are preferential nose breathers who only resort to oral or oronasal breathing under the demanding conditions of exercise, nasal obstruction, or speech. This occurs despite the fact that upper airway resistance via the nasal airway is about twice that via a mouthpiece. However, Bethel et al. (1983b) suggest more asthmatics may breath oronasally and that asthmatics switch from nasal to oronasal breathing at a lower ventilation than normals; this is due to the greater prevalence of rhinitis in the asthmatic population.

4.4.3 Tolerance (Attenuation of Response) to SO_2 With Repeated Exposure

Attenuation of SO_2-induced bronchoconstriction with repeated SO_2 exposure (with eucapnic hyperpnea) was not associated with a decrease in airway responsiveness to histamine (Sheppard et al., 1983). This probably indicates that the attenuation of response was not related to decreased responsiveness of airway smooth muscle or decreased responsiveness of vagal reflex pathways. However it is possible that, within the lung, the regional dosimetry of the gas, SO_2, and the aerosol, histamine, were quite different. If the changes in response to repeated SO_2 exposure were due to localized effects, the histamine aerosol may have been an inappropriate probe. These authors did suggest that depletion of mediators or a selective inhibition of SO_2-sensitive afferents might be involved in this phenomenon. For equivalent total exercise time, Kehrl et al. (1986) observed greater SO_2-induced bronchoconstriction with continuous as compared to intermittent exercise during SO_2 exposure. Thus mediator depletion or selective inhibition of afferents, as well as exercise-induced release of endogenous bronchodilators (epinephrine) are probably not related to the attenuation of response with repeated exposure (or repeated intermittent exercise during exposure). These results suggest that alleviation of SO_2-induced bronchoconstriction is related to events that occur during the post-exposure/post-exercise recovery period.

Attenuation of bronchoconstriction has been reported for exercise (Stearns et al., 1981) and hyperpnea of cold, dry air (Bar-Yishay et al. 1983, Wilson et al., 1982) repeated at short time intervals, suggesting that the attenuation of SO_2-induced bronchoconstriction may be secondary to this decline in response.

4.5 SUMMARY AND CONCLUSIONS

Studies which have been published in the scientific literature since 1982 support many of the conclusions reached in the earlier criteria document (U.S. EPA, 1982) and the previous addendum (U.S. EPA, 1982c).

The new studies clearly demonstrate that asthmatics are much more sensitive to SO_2 as a group. Nevertheless, it is clear that there is a broad range of sensitivity to SO_2 among asthmatics exposed under similar conditions. Recent studies also confirm that normal healthy subjects, even with moderate to heavy exercise, do not experience effects on pulmonary function due to SO_2 exposure in the range of 0 to 2 ppm. The minor exception may be the annoyance of the unpleasant smell or taste associated with SO_2. The suggestion that asthmatics are about an order of magnitude more sensitive than normals is thus confirmed.

There is no longer any question that normally breathing asthmatics performing moderate to heavy exercise will experience SO_2-induced bronchoconstriction when breathing SO_2 for at least 5 min at concentrations less than 1 ppm. Durations beyond 10 min do not appear to cause substantial worsening of the effect. The lowest concentration at which bronchoconstriction is clearly worsened by SO_2 breathing depends on a variety of factors.

Exposure to less than 0.25 ppm has not evoked group mean changes in responses. Although some individuals may appear to respond to SO_2 concentrations less than 0.25 ppm, the frequency of these responses is not demonstrably greater than with clean air. Thus individual responses cannot be relied upon for response estimates, even in the most reactive segment of the population.

In the SO_2 concentration range from 0.2 to 0.3 ppm, six chamber exposure studies were performed with asthmatics performing moderate to heavy exercise. The evidence that SO_2-induced bronchoconstriction occurred at this concentration with natural breathing under a range of ambient conditions was equivocal. Only with oral mouthpiece breathing of dry air (an unusual breathing mode under exceptional ambient conditions) were small effects observed on a test of questionable quantitative relevance for criteria development purposes. These findings are in accord with the observation that the most reactive subject in the Horstman et al. (1986) study had a $PCSO_2$ (SO_2 concentration required to double SRaw) of 0.28 ppm.

Several observations of significant group mean changes in SRaw have recently been reported for asthmatics exposed to 0.4 to 0.6 ppm SO_2. Most if not all studies, using moderate to heavy exercise levels (>40 to 50 L/min), found evidence of bronchoconstriction at 0.5 ppm. At a lower exercise rate, other studies (e.g., Schachter et al., 1984) did not produce clear evidence of SO_2-induced bronchoconstriction at 0.5 ppm SO_2. Exposures which included higher ventilations, mouthpiece breathing, and inspired air with a low water content resulted in the greatest responses. Mean responses ranged from 45 percent (Roger et al., 1985) to 280 percent (Bethel et al., 1983b) increase in SRaw. At concentrations in the range of 0.6 to 1.0 ppm, marked increases in SRaw are observed following exposure. Recovery is generally complete within approximately 1 h although the recovery period may be longer for subjects with the most severe responses.

It is now evident that for SO_2-induced bronchoconstriction to occur in asthmatics at concentrations less than 0.75 ppm, the exposure must be accompanied by hyperpnea. Ventilations in the range of 40 to 60 L/min have been most appropriate; such ventilations are beyond the usual oronasal ventilatory switchpoint. There is no longer any question that oral breathing (especially via mouthpiece) causes exacerbation of SO_2-induced bronchoconstriction. New studies reinforce the concept that the mode of breathing is an important determinant of the intensity of SO_2-induced bronchoconstriction in the following order: oral > oronasal > nasal. A second exacerbating factor implicated in recent reports is the breathing of dry and/or cold air. It is not clearly established whether the exacerbation of the SO_2 effects is due to airway cooling, airway drying, or some other mechanism.

The new studies do not provide sufficient additional information to establish whether the intensity of the SO_2-induced bronchoconstriction depends upon the severity of the disease. Across a broad clinical range from "normal" to moderate asthmatic there is clearly a relationship between the presence of asthma and sensitivity to SO_2. Within the asthmatic population, the relationship of SO_2 sensitivity to the qualitative clinical severity of asthma has not been studied systematically. Ethical considerations (i.e., continuation of appropriate medical treatment) prevent the unmedicated exposure of the "severe" asthmatic because of his dependence upon drugs for control of his asthma. True determination of sensitivity requires that the interference with SO_2 response caused by such medication be removed. Because of these mutually exclusive requirements, it is unlikely that the "true" SO_2 sensitivity of severe asthmatics will be determined. Nevertheless, more severe asthmatics should be studied. Alternative methods to those used with mild asthmatics, not critically dependant on regular medication, will be required. The studies to date have only addressed the "mild to moderate" asthmatic.

Consecutive SO_2 exposures (repeated within 30 min or less) result in a diminished response compared with the initial exposure. It is apparent that this refractory period lasts at least 30 min but that normal reactivity returns within 5 h. The mechanisms and time course of this effect are not clearly established but refractoriness does not appear to be related to an overall decrease in bronchomotor responsiveness. These observations suggests that the effects of SO_2 on airway resistance and spirometry tend to be short lasting and do not tend to become worse with continued or repeated exposure. Nevertheless, the issue of chronic exposure to SO_2 in asthmatics has not been addressed.

From the review of studies included in this addendum, it is clear that the magnitude of response (typically bronchoconstriction) induced by any given SO_2 concentration was variable among individual asthmatics. Exposures to SO_2 concentrations of 0.25 ppm or less, which did not induce significant group mean increases in airway resistance also did not cause symptomatic bronchoconstriction in individual asthmatics. On the other hand, exposures to 0.40 ppm SO_2 or greater (combined with moderate to heavy exercise) which induced significant group mean increases in airway resistance,also caused substantial bronchoconstriction in some individual asthmatics. This bronchoconstriction was often associated with wheezing and the perception of respiratory distress. In a few instances it was necessary to discontinue the exposure and provide medication. The significance of these observations is that some SO_2-sensitive asthmatics are at risk of experiencing clinically significant (i.e., symptomatic) bronchoconstriction requiring termination of activity and/or medical intervention when exposed to SO_2 concentrations of 0.40 to 0.50 ppm or greater when this exposure is accompanied by at least moderate activity.

CHAPTER 5. EXECUTIVE SUMMARY

In general, studies published in the scientific literature since 1981-82 support many of the conclusions reached in the earlier criteria review (U.S. EPA, 1982a,c). Some of the key findings emerging from the present evaluation of the newly available information on health effects associated with exposure to PM and SO_x are summarized here.

5.1 RESPIRATORY TRACT DEPOSITION AND FATE

Studies published since preparation of the earlier criteria document (U.S. EPA, 1982a) and the previous addendum (U.S. EPA, 1982c) support the conclusions reached at that time and provide clarification of several issues. In light of previously available data, new literature was reviewed with a focus towards (1) the thoracic deposition and clearance of large particles, (2) assessment of deposition during oronasal breathing, (3) deposition in possibly susceptible subpopulations, such as children, and (4) information that would relate the data to refinement or interpretation of ancillary issues, such as inter- and intrasubject variability in deposition, deposition of monodisperse versus polydisperse aerosols, etc.

The thoracic deposition of particles ≥ 10 μm D_{ae} and their distribution in the TB and P regions has been studied by a number of investigators (Svartengren, 1986; Heyder, 1986; Emmett et al., 1982). Depending upon the breathing regimen used, TB deposition ranged from 0.14 to 0.36 for 10-μm D_{ae} particles, while the range for 12-μm D_{ae} particles was 0.09 to 0.27. For particles 16.4 μm D_{ae}, a maximally deep inhalation pattern resulted in TB deposition of 0.12. While the magnitude of deposition in various regions depends heavily upon minute ventilation, there is, in general, a gradual decline in thoracic deposition for large particle sizes, and there can be significant deposition of particles greater than 10 μm D_{ae}, particularly for individuals who habitually breathe through their mouth. Thus, the deposition experiments wherein subjects inhale through a mouthpiece are relevant to examining the potential of particles to penetrate to the lower respiratory tract and pose a potentially increased risk. Increased risk may be due to increased localized dose or to the exceedingly long half-times for clearance of larger particles (Gerrity et al., 1983).

Although experimental data are not currently available for deposition of particles in the lungs of children, some trends are evident from the modeling results of Phalen et al. (1985). Phalen and co-workers made morphometric measurements in replica lung casts of people aged 11 days to 21 years and modeled deposition during inspiration as a function of activity level. They found that, in general, increasing age is associated with decreasing particulate tracheobronchial deposition efficiency. However, very high flow rates and large particulate sizes do not exhibit consistent age-dependent differences. Since minute ventilation at a given state of activity is approximately linearly related to body mass, children receive a higher TB dose of particles than do adults and would appear to be at a greater risk, other factors (i.e., mucociliary clearance, particulate losses in the head, tissue sensitivity, etc.) being equal.

5.2 SUMMARY OF EPIDEMIOLOGIC FINDINGS ON HEALTH EFFECTS ASSOCIATED WITH EXPOSURE TO AIRBORNE PARTICLES AND SO_X

Newly available reanalyses of data relating mortality in London to short-term (24-h) exposures to PM (measured as smoke) and SO_2 were evaluated and their results compared with earlier findings and conclusions discussed in U.S. EPA (1982a). Varying strengths and weaknesses were evident in relation to the different individual reanalyses (Mazumdar et al., 1982; Ostro, 1984; Shumway et al., 1983; Schwartz and Marcus, 1986) evaluated and certain issues remain unresolved. Nevertheless, the following conclusions appear to be warranted based on the earlier criteria review (U.S. EPA, 1982a) and present evaluation of newly available analyses of the London mortality experience: (1) markedly increased mortality occurred, mainly among the elderly and chronically ill, in association with BS and SO_2 concentrations above 1000 $\mu g/m^3$, especially during episodes when such pollutant elevations occurred for several consecutive days; (2) during such episodes coincident high humidity (fog) was also likely important, possibly in providing conditions leading to formation of H_2SO_4 or other acidic aerosols; (3) increased risk of mortality is associated with exposure to BS and SO_2 levels in the range of 500 to 1000 $\mu g/m^3$, for SO_2 most clearly at concentrations in excess of ~700 to 750 $\mu g/m^3$; and (4) convincing evidence indicates that relatively small but statistically significant increases in the risk of mortality exist at BS (but not SO_2 levels below 500 $\mu g/m^3$), with no indications of any specific threshold level having yet been demonstrated at lower concentrations of BS (e.g., at 150 $\mu g/m^3$). However, precise quantitative specification of the lower PM levels associated with mortality is not possible, nor can one rule out potential contributions of other possible confounding variables at these low PM levels.

In addition to the reanalyses of London mortality data, reanalyses of mortality data from New York City in relation to air pollution reported by Ozkaynak and Spengler (1985) were evaluated. Time-series analyses were carried out on a subset of New York City data included in a prior analysis by Schimmel (1978) which was critiqued during the earlier criteria review (U.S. EPA, 1982a). The reanalyses by Ozkaynak and Spengler (1985) evaluated 14 years (1963-76) of daily measurements of mortality (the sum of heart, other circulatory, respiratory, and cancer mortality), COH, SO_2, and temperature. In summary, the newly available reanalyses of New York City data raise possibilities that, with additional work, further insights may emerge regarding mortality-air pollution relationships in a large U.S. urban area. However, the interim results reported thus far do not now permit definitive determination of their usefulness for defining exposure-effect relationships, given the above-noted types of caveats and limitations.

Similarly, it is presently difficult to accept findings reported in another new study (Hazakis et al., 1986) of mortality associated with relatively low levels of SO_2 pollution in Athens, given questions regarding representativeness of the monitoring data and the statistical soundness of using deviations of mortality from an earlier baseline relatively distant in time. Lastly, a newly reported study (Mazumdar and Sussman, 1983) of mortality-air pollution relationships in Pittsburgh (Allegheny County, PA) was evaluated as having utilized inadequate exposure characterization and the results contain sufficient internal inconsistencies, so that the analyses are not useful for delineating mortality relationships with either SO_2 or PM.

Of the newly-reported analyses of short-term PM/SO_x exposure-morbidity relationships discussed in this Addendum, the Dockery et al. (1982) study provides the best-substantiated and most readily interpretable results. Those results, specifically, point toward decrements in lung function occurring in association with acute, short-term increases in PM and SO_2 air pollution. The small, reversible decrements appear to persist for 2-3 wks after episodic exposures to these pollutants across a wide range of concentrations, with no clear delineation of threshold yet being evident. In some study periods effects may have been due to 24-hr TSP and SO_2 levels ranging up to 220-420 and 280-460 $\mu g/m^3$, respectively. Notably larger decrements in lung function were discernable for a subset of children (responders) than for others. The precise medical significance of the observed decrements per se or any consequent long-term sequelae remain to be determined. The nature and magnitude of lung function decrements found by Dockery et al. (1982), it should be noted, are also consistent with: (1) the recently reported findings of Dassen et al (1986) of pulmonary function decrements of approximately the same magnitude over similar time periods after episodic exposure of Dutch children to 24-hr TSP and SO_2 levels in the 200-250 $\mu g/m^3$ range, (2) observations of Stebbings and Fogleman (1979) of gradual recovery in lung function of children during seven days following a high PM episode in Pittsburgh, PA (max 1-hr TSP estimated at 700 $\mu g/m^3$); and (3) a report by Saric et al. (1981) of 5 percent average declines in $FEV_{1.0}$ being associated with high SO_2 days (89-235 $\mu g/m^3$).

In regard to evaluation of long-term exposure effects, the 1982 U.S. EPA criteria document (1982a) noted that certain large-scale "macroepidemiological" (or "ecologic") studies have attracted attention on the basis of reported demonstrations of associations between mortality and various indices of air pollution, e.g., PM or SO_x levels. U.S. EPA (1982a) also noted that various criticisms of then-available ecologic studies made it impossible to ascertain which findings may be more valid than others. Thus, although many of the studies qualitatively suggested positive associations between mortality and chronic exposure to certain air pollutants in the United States, many key issues remained unresolved concerning reported associations and whether they were causal or not.

Since preparation of the earlier Criteria Document (U.S. EPA, 1982a) additional ecological analyses have been reported regarding efforts to assess relationships between mortality and long-term exposure to particulate matter and other air pollutants. For example, Lipfert (1984) conducted a series of cross-sectional multiple regression analyses of 1969 and 1970 mortality rates for up to 112 U.S. SMSA's, using the same basic data set as Lave and Seskin

(1977) for 1969 and taking into account various demographic, environmental and lifestyle variables (e.g., socioeconomic status and smoking). Also, the Lipfert (1984) reanalysis included several additional independent variables: diet; drinking water variables; use of residential heating fuels; migration; and SMSA growth. New dependent variables included age-specific mortality rates with their accompanying sex-specific age variables. Both linear and several nonlinear (e.g., quadratic or linear splines testing for possible threshold model specifications) were evaluated.

It became quite evident from the results obtained that the air pollution regression results for the U.S. data sets analyzed by Lipfert (1984) are extremely sensitive to variations in the inclusion/exclusion of specific observations (for central city versus SMSA's or different subsets of locations) or additional explanatory variables beyond those used in the earlier Lave and Seskin analyses. The results are also highly dependent upon the particular model specifications used, i.e. air pollution coefficients vary in strength of association with total or age-/sex-specific mortality depending upon the form of the specification and the range of explanatory variables included in the analyses. Lipfert's overall conclusion was that the sulfate regression coefficients are not credible and, since sulfate and TSP interact with each other in these regressions, caution is warranted for TSP coefficients as well.

Ozkaynak and Spengler (1985) have also newly described results from ongoing attempts to improve upon previous analyses of mortality and morbidity effects of air pollution in the United States. Ozkaynak and Spengler (1985) present principal findings from a cross-sectional analysis of the 1980 U.S. vital statistics and available air pollution data bases for sulfates, and fine, inhalable and total suspended particles. In these analyses, using multiple regression methods, the association between various particle measures and 1980 total mortality were estimated for 98 and 38 SMSA subsets by incorporating recent information on particle size relationships and a set of socioeconomic variables to control for potential confounding. Issues of model misspecification and spatial autocorrelation of the residuals were also investigated.

The Ozkaynak and Spengler (1985) results for 1980 U.S. mortality provide an interesting overall contrast to the findings of Lipfert (1984) for 1969-70 U.S. mortality data. Whereas Lipfert found TSP coefficients to be most consistently statistically significant (although varying widely depending upon model specifications, explanatory variables included, etc.), Ozkaynak and

Spengler found particle mass measures including coarse particles (TSP, IP) often to be nonsignificant predictors of total mortality. Also, whereas Lipfert found the sulfate coefficients to be even more unstable than the TSP associations with mortality (and questioned the credibility of the sulfate coefficients), Ozkaynak and Spengler found that particle exposure measures related to the respirable or toxic fraction of the aerosols (e.g., FP or sulfates) to be most consistently and significantly associated with annual cross-sectional mortality rates. It might be tempting to hypothesize that changes in air quality or other factors from the earlier data sets (for 1969-70) analyzed by Lipfert (1984) to the later data (for 1980) analyzed by Ozkaynak and Spengler (1985) and Ozkaynak et al. (1986) may at least partly explain their contrasting results, but there is at present no basis by which to determine if this is the case or which set of findings may or may not most accurately characterize associations between mortality and chronic PM or SO_x exposures in the United States. Thus conclusions stated in U.S EPA (1982a) concerning ecologic analyses still largely apply in regard to mortality-PM/SO_x relationships.

The present Addendum also evaluated a growing body of new literature on morbidity effects associated with chronic exposures to airborne particles and sulfur oxides. In summary, of the numerous new studies published on morbidity effects associated with long-term exposures to PM or SO_x, only a few may provide potentially useful results by which to derive quantitative conclusions concerning exposure-effect relationships for the subject pollutants. A study by Ware et al. (1986), for example, provides evidence of respiratory symptoms in children being associated with particulate matter exposures in contemporary U.S. cities without evident threshold across a range of annual-average TSP levels of ~30 to 150 $\mu g/m^3$, with most marked effects notable in the 60-150 $\mu g/m^3$ range in comparison to lower TSP levels. The increase in symptoms appears to occur without concomitant decrements in lung function among the same children. The medical significance of the observed increases in symptoms unaccompanied by decrements in lung function remains to be fully evaluated but is of likely health concern. Caution is warranted, however, in using these findings for risk assessment purposes in view of the lack of significant associations for the same variables when assessed from data within individual cities included in the Ware et al. (1986) study. The findings derived from another series of studies (Ostro, 1983; Hausman et al., 1984; Ostro, 1987) are qualitatively indicative of morbidity effects in adults being associated with PM exposures over time within U.S. cities, and these results tend to support the plausibility that the associations found by Ware et al. (1986) reflect actual morbidity effects in children due to contemporaneous U.S. PM exposures.

Other new American studies provide evidence for: (1) increased respiratory symptoms among young adults in association with annual-average SO_2 levels of ~115 $\mu g/m^3$ (Chapman et al., 1985); and (2) increased prevalence of cough in children (but not lung function changes) being associated with intermittent exposures to mean peak 3-hr SO_2 levels of ~1.0 ppm or annual average SO_2 levels of ~103 $\mu g/m^3$ (Dodge et al., 1985). It is difficult to distinguish as to whether the effects found in these two studies are due to repeated high-level SO_2 peak exposures or to chronic exposures to lower concentrations of SO_2 or its transformation products.

Results from one European study (PAARC, 1982a,b) also tend to suggest the likelihood of lower respiratory disease symptoms and decrements in lung function in adults (both male and female) being associated with annual average SO_2 levels ranging without evident threshold from about 25 to 130 $\mu g/m^3$. In addition that study suggests that upper respiratory disease and lung function decrements in children may also be associated with annual-average SO_2 levels across the above range. However, the associations between morbidity effects and SO_2 reported by PAARC (1982a,b) cannot be fully accepted due to: (1) internal inconsistencies between results obtained with SO_2 exposure estimates based on one type of measurement method versus those based on another SO_2 measurement technique, and (2) the lack of adequate control for seasonal effects and parental smoking in certain analyses for childrens' data that yielded significant health effects associations.

5.3 SUMMARY OF CONTROLLED HUMAN EXPOSURE STUDIES OF SULFUR DIOXIDE HEALTH EFFECTS

The new studies evaluated in the present addendum (Chapter 4) clearly demonstrate that asthmatics are much more sensitive to SO_2 as a group. Nevertheless, it is clear that there is a broad range of sensitivity to SO_2 among asthmatics exposed under similar conditions. Recent studies also confirm that normal healthy subjects, even with moderate to heavy exercise, do not experience effects on pulmonary function due to SO_2 exposure in the range of 0 to 2 ppm. The minor exception may be the annoyance of the unpleasant smell or taste associated with SO_2. The suggestion that mild "compensated" asthmatics are about an order of magnitude more sensitive than normals is thus confirmed. There is not enough information on SO_2 response in moderate to severe asthmatics to estimate their sensitivity.

There is no longer any question that normally breathing asthmatics performing moderate to heavy exercise will experience SO_2-induced bronchoconstriction when breathing SO_2 for at least 5 min at concentrations less than 1 ppm. Durations beyond 10 min do not appear to cause substantial worsening of the effect. The lowest concentration at which bronchoconstriction is clearly exacerbated by SO_2 breathing depends on a variety of factors.

Exposure to less than 0.25 ppm has not evoked group mean changes in responses. Although some individuals may appear to respond to SO_2 concentrations less than 0.25 ppm, the frequency of these responses is not demonstrably greater than with clean air. Thus individual responses cannot be relied upon for response estimates, even in the most reactive segment of the population.

In the SO_2 concentration range from 0.2 to 0.3 ppm, six chamber exposure studies were performed with asthmatics performing moderate to heavy exercise. The evidence that SO_2-induced bronchoconstriction occurred at this concentration with natural breathing under a range of ambient conditions was equivocal. Only with oral mouthpiece breathing of dry air (an unusual breathing mode under exceptional ambient conditions) were small effects observed on a test of questionable quantitative relevance for criteria development purposes. These findings are in accord with the observation that the most reactive subject in the Horstman et al. (1986) study had a $PCSO_2$ (SO_2 concentration required to double SRaw) of 0.28 ppm.

Several observations of significant group mean changes in SRaw have recently been reported for asthmatics exposed to 0.4 to 0.6 ppm SO_2. Most if not all studies, using moderate to heavy exercise levels (>40 to 50 L/min), found evidence of bronchoconstriction at 0.5 ppm. At a lower exercise rate, other studies (e.g., Schachter et al., 1984) did not produce clear evidence of SO_2-induced bronchoconstriction at 0.5 ppm SO_2. Exposures which included higher ventilations, mouthpiece breathing, and inspired air with a low water content resulted in the greatest responses. Mean responses ranged from 45 percent (Roger et al., 1985) to 280 percent (Bethel et al., 1983b) increase in

SRaw. At concentrations in the range of 0.6 to 1.0 ppm, marked increases in SRaw are observed following exposure. Recovery is generally complete within approximately 1 h although the recovery period may be longer for subjects with the most severe responses.

It is now evident that for SO_2-induced bronchoconstriction to occur in asthmatics at concentrations less than 0.75 ppm, the exposure must be accompanied by hyperpnea. Ventilations in the range of 40 to 60 L/min have been most appropriate; such ventilations are beyond the usual oronasal ventilatory switchpoint. There is no longer any question that oral breathing (especially via mouthpiece) causes exacerbation of SO_2-induced bronchoconstriction. New studies reinforce the concept that the mode of breathing is an important determinant of the intensity of SO_2-induced bronchoconstriction in the following order: oral > oronasal > nasal. A second exacerbating factor strongly implicated in recent reports is the breathing of dry and/or cold air with SO_2. It is not established whether the reduced water content, the reduced temperature, or both is responsible for this effect.

The new studies do not provide sufficient additional information to establish whether the intensity of the SO_2-induced bronchoconstriction depends upon the severity of the disease. Across a broad clinical range from "normal" to moderate asthmatic there is clearly a relationship between the presence of asthma and sensitivity to SO_2. Within the asthmatic population, the relationship of SO_2 sensitivity to the qualitative clinical severity of asthma has not been studied systematically. Ethical considerations (i.e., continuation of appropriate medical treatment) prevent the unmedicated exposure of the "severe" asthmatic because of his dependence upon drugs for control of his asthma. True determination of sensitivity requires that the interference with SO_2 response caused by such medication be removed. Because of these mutually exclusive requirements, it is unlikely that the "true" SO_2 sensitivity of severe asthmatics will be determined. Nevertheless, more severe asthmatics should be studied. Alternative methods to those used with mild asthmatics, not critically dependant on regular medication, will be required. The studies to date have only addressed the "mild to moderate" asthmatic.

Consecutive SO_2 exposures (repeated within 30 min or less) result in a diminished response compared with the initial exposure. It is apparent that this refractory period lasts at least 30 min but that normal reactivity returns within 5 h. The mechanisms and time course of this effect are not clearly established but refractoriness does not appear to be related to an overall decrease in bronchomotor responsiveness.

From the review of studies included in this addendum, it is clear that the magnitude of response (typically bronchoconstriction) induced by any given SO_2 concentration was variable among individual asthmatics. Exposures to SO_2 concentrations of 0.25 ppm or less, which did not induce significant group mean increases in airway resistance also did not cause symptomatic bronchoconstriction in individual asthmatics. On the other hand, exposures to 0.40 ppm SO_2 or greater (combined with moderate to heavy exercise) which induced significant group mean increases in airway resistance, also caused substantial bronchoconstriction in some individual asthmatics. This bronchoconstriction was often associated with wheezing and the perception of respiratory distress. In a few instances it was necessary to discontinue the exposure and provide medication. The significance of these observations is that some SO_2-sensitive asthmatics are at risk of experiencing clinically significant (i.e., symptomatic) bronchoconstriction requiring termination of activity and/or medical intervention when exposed to SO_2 concentrations of 0.40 to 0.50 ppm or greater when this exposure is accompanied by at least moderate activity.

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APPENDIX

Schwartz and Marcus Statistical Reanalysis of Mortality Data during 14 London Winters (1958-1972).

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
Office of Air Quality Planning and Standards
Research Triangle Park, North Carolina 27711

October 14, 1986

MEMORANDUM

SUBJECT: Statistical Reanalyses of Data Relating Mortality to Air Pollution During 14 London Winters (1958-1972)

FROM: Allan Marcus Allan H Marcus
Ambient Standards Branch

Joel Schwartz Joe
Economic and Regulatory Analysis Division

TO: Bruce Jordan, Chief
Ambient Standards Branch

Les Grant, Director
Environmental Criteria and Assessment Office

This memo summarizes our continuing analysis of the London mortality data. These analyses were conducted at your request for the purpose of delineating further the degree of reliance that can be put on the more recent published analyses of these data discussed in the criteria document and staff paper addenda. Our analyses are discussed in more detail in the attached paper. The recently published studies include three statistical analyses of the possible relationship between daily air pollution concentrations and days with excessive numbers of deaths in London during the winters of 1958-1972. (Mazumdar et al., 1982; Ostro, 1984; Shumway et al., 1983)

We believe that these studies have shown that a relationship exists between particulate matter as measured by British Smoke or SO_2 and mortality in London, and that those relationships continue below a British Smoke level of 150 $\mu g/m^3$. However, commenters and others have raised questions about 1) whether the analyses adequately handled the temporal structure of the data, both in terms of avoiding confounding due to long term time trends and seasonal fluctuations, and in terms of avoiding the mis-estimation of the regression standard errors (and hence significance tests) that occurs when there is autocorrelation in the regression residuals; 2) whether the dose response relationship is linear or nonlinear and whether that relationship is distorted by the techniques used to filter the series, and 3) whether it is British Smoke, SO_2 or both that are responsible for the mortality.

All of the studies have attempted to deal with the autocorrelation in the data (that is, the number of people who die on day t is correlated with the number of people who died on day t-1, t-2, etc.). Some used deviations from 15 day moving average to remove these autocorrelations, more recently Ostro used an autoregressive model. All of the studies used separate regressions for each year to remove the time trend of falling mortality. None of the models successfully separated SO_2 and British Smoke effects. None of the studies reported any tests to determine whether using separate regressions for each year adequately dealt with the possibility of a linear time trend in the data or reported any formal tests to determine whether they had in fact adequately accounted for the autocorrelations in the data. In addition, while the studies clearly demonstrated relationships at low levels, and gave some indication of their magnitude, no detailed exploratory analysis was presented to determine the potential shape of the dose response curve.

With the assistance of analysts at the California Air Resources Board, who had obtained the full 14 winter data set from the United Kingdom, we therefore decided to reanalyze the London mortality data so as to evaluate the adequacy of the fitted models. We began by grouping and examining the raw data graphically for all years and each year separately much in the manner used by Ware et al. (1981) and the 1982 Criteria Document for the 1958-59 winter. Figure 1 shows the results of raw daily mortality versus smoke for all winters while Figure 2 shows the same plot for all days with smoke less than 500 $\mu g/m^3$. Figure 3 illustrates deviations in daily mortality for a representative individual winter. In general, these plots show the same kind of continuum of association between mortality and smoke seen in the earlier analyses, with no apparent lower limit. Both the curvilinear shape of the dose response curve and the low level effects are also evident in year by year plots. We then decided to study the temporal structure of the process, particularly its autocorrelation. We developed regression models that control for the effects of autocorrelation. These models were then used to study the relative usefulness of BS and SO_2 as predictors of mortality. A comparison of the key features of these as well as the published regressions is summarized in Table 1.

These additional analyses have suggested the following conclusions:

(A) Short-term changes in mortality can be very well modeled by an autoregressive process with two or three terms (i.e., mortality on day t predicted by a combination of residual mortality on days t-1, t-2, and possibly t-3). The autoregressive part (AR1-3) alone usually accounted for about 54% of the variance in each year's daily mortality. When only days with BS < 200 were considered, the fraction of variance explained by autoregression increased to about 58%. The AR3 structure of the data was not completely modeled by either the 15-day average detrending or by the AR1 model used in an unpublished analysis by Ostro.

(B) When temperature, humidity and one pollutant were considered in an autoregressive model, the incremental variation in mortality explained was about 14%. Using British Smoke as the exposure variable, pollution was significantly related to mortality in 13 out of 14 years. In a random effects model that combined the results from all years British Smoke was highly significant (P < .0001). When smoke levels were restricted to only those below 500 μg/m^3 or even below 200 μg/m^3 the overall significance of British Smoke increased ($t = 8.74$ for levels below 500 μg/m^3 and $t = 14.43$ for levels below 200 μg/m^3).

(C) Regression models for mortality using BS, SO_2, temperature and humidity accounted for substantial additional variance in daily mortality over and above the autoregressive components. This regression model added an additional fraction of explained variance, about 14%, to the autoregressive model. Even when only days with BS < 200 were considered the regression model explained about 12% of the variance in mortality.

(D) Air pollution variables were usually more significant statistically than temperature or humidity in explaining mortality.

(E) Due to the multicollinearity of BS and SO_2, there were no years when both were significant at the 5% level of significance. However, BS was always significant statistically for more years than was SO_2. Using all data, BS was significant in 6 years and SO_2 in 2 years out of 14, and even on days with BS < 500, BS was significant 4 years and SO_2 was never significant in 14 years when both variables were used. These results are shown in Table 2. However, the statistical significance of both variables was greatly reduced because of the multicollinearity. What is more striking is that the multiple regression slope for BS was relatively stable whether or not SO_2 was used in the model, with a mean slope (weighted by the reciprocal of the variance) of 0.079 excess deaths per μg/m^3 BS without SO_2 in the model, and 0.061 with SO_2 included. The estimated slopes for SO_2 were significantly positive only for 1958 and 1962, and were otherwise insignificant and scattered around zero slope. Even on days with BS < 200 only, the mean yearly slopes for BS were 0.138 without SO_2 and 0.135 with SO_2 included (both highly significant) (Tables 3,4). Again, the SO_2 slope was approximately zero.

Our recent assessments thus confirm that the multiple regression models for daily mortality and BS do indeed reflect a relationship that cannot be attributed to time series effects, temperature, SO_2, or functional misspecification. The general consistency of the results is shown in Figure 4. This shows the regression slope of mortality vs. BS for Mazumdar's linear model (coded M), for Ostro's low-BS linear model (BS < 150, coded L), and for the CARB/EPA analyses using all days (coded A), days with BS < 500 (coded B), and days with BS < 200 (coded D). There does appear to be some tendency for higher slopes in later years when both BS and SO_2 levels reflect more nearly contemporary conditions. Thus the published analyses do appear to be relevant in assessing the health effects of particulate matter.

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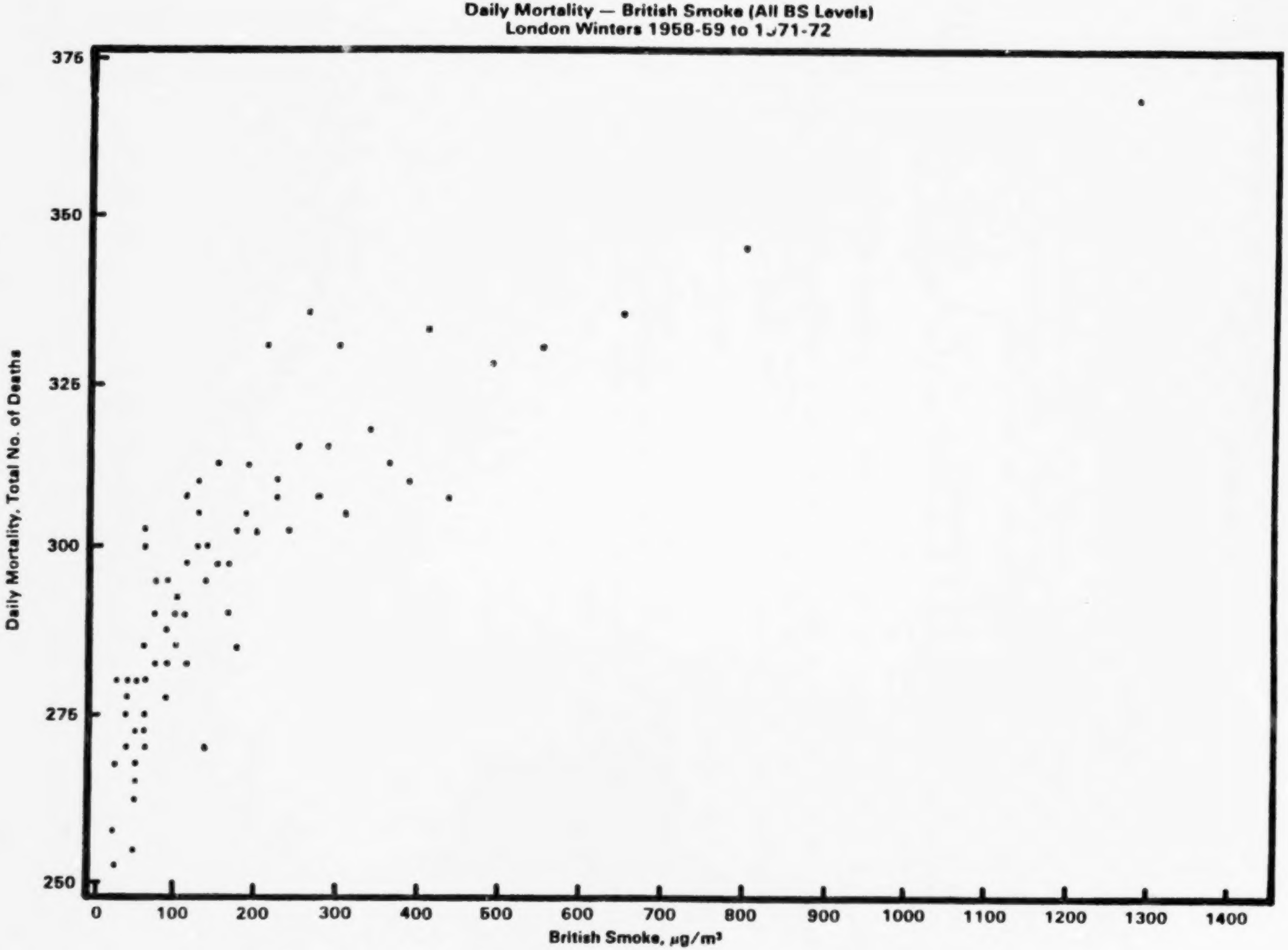

Figure 1. Daily mortality in London during the winters of 1958-1959 through 1971-1972. Each point represents the average daily mortality and average BS for 20 adjacent values of BS. Note the data suggest a higher slope at lower levels of BS.

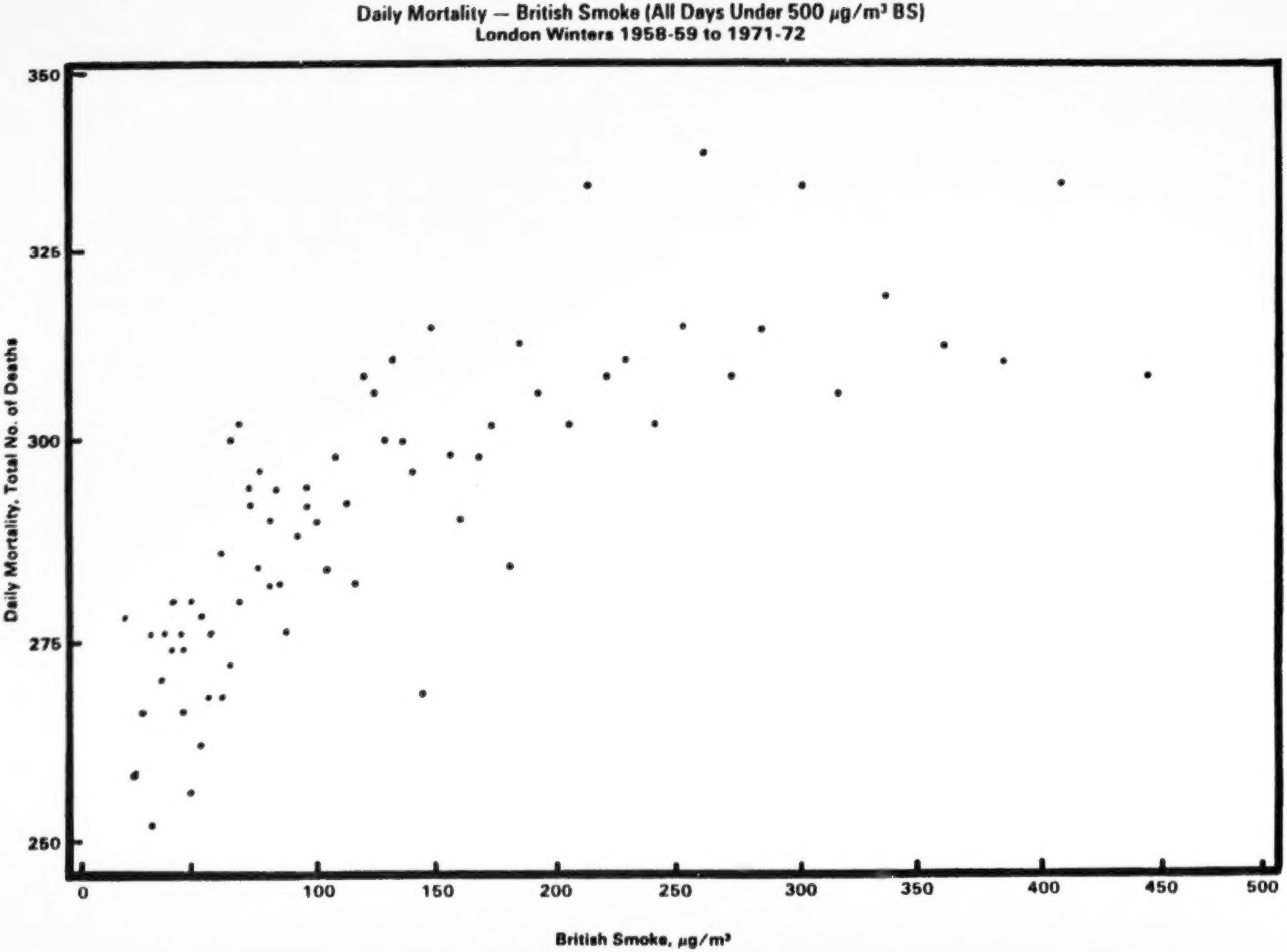

Figure 2. Daily mortality in London during the winters of 1958-1959 through 1971-1972. Each point represents the average daily mortality and average BS for 20 adjacent values of BS. The curvilinear relationship is still apparent, and the relationship clearly continues below BS levels of 100 μg/m^3.

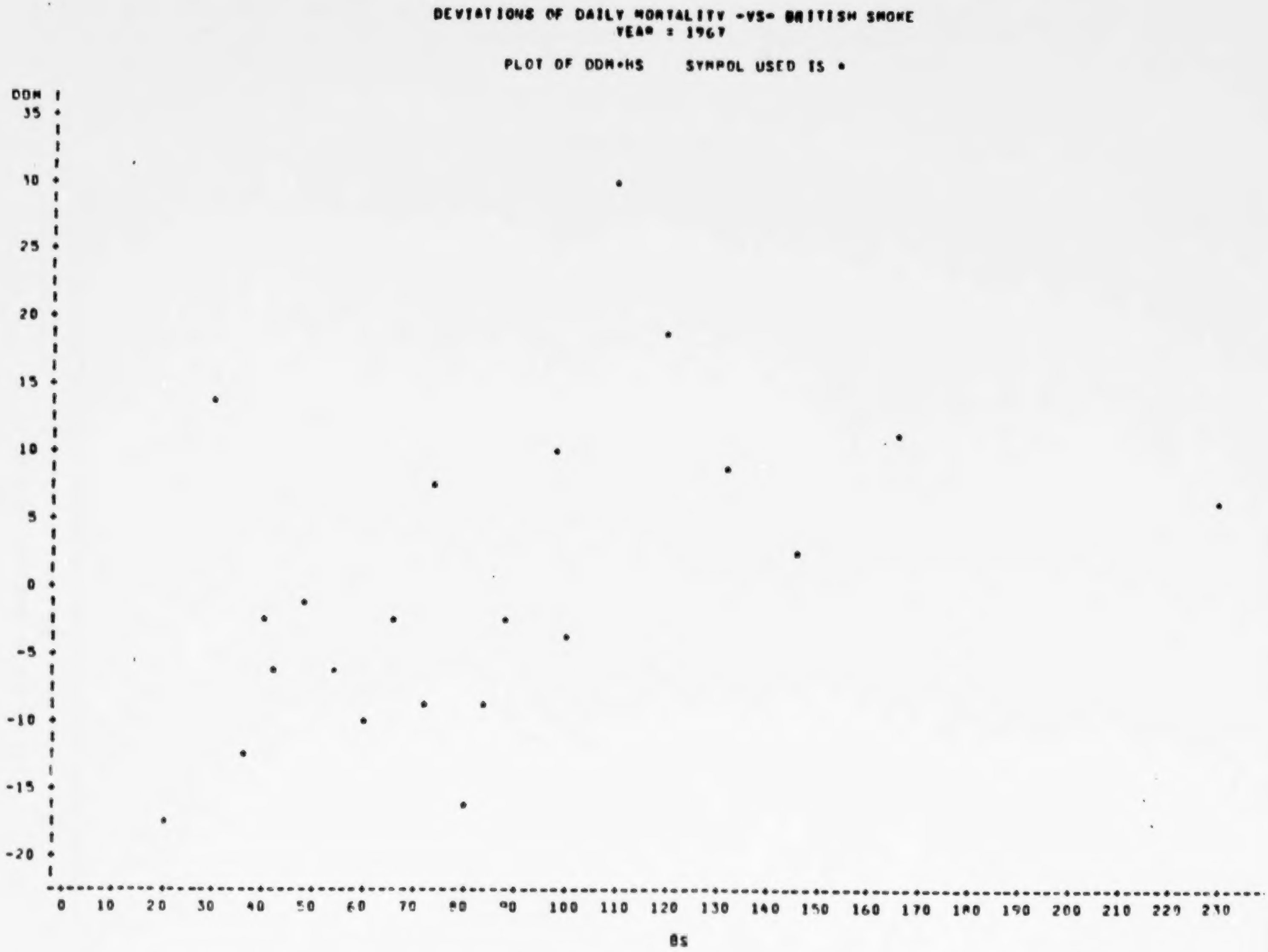

Figure 3. Deviations of daily mortality from 75-day moving average of mortality for the London winter of 1968-1969. Each point represents the average daily mortality and average BS for 5 adjacent values of BS. Deviation in daily mortality shows a relationship to BS similar to that for total mortality, again without apparent threshold.

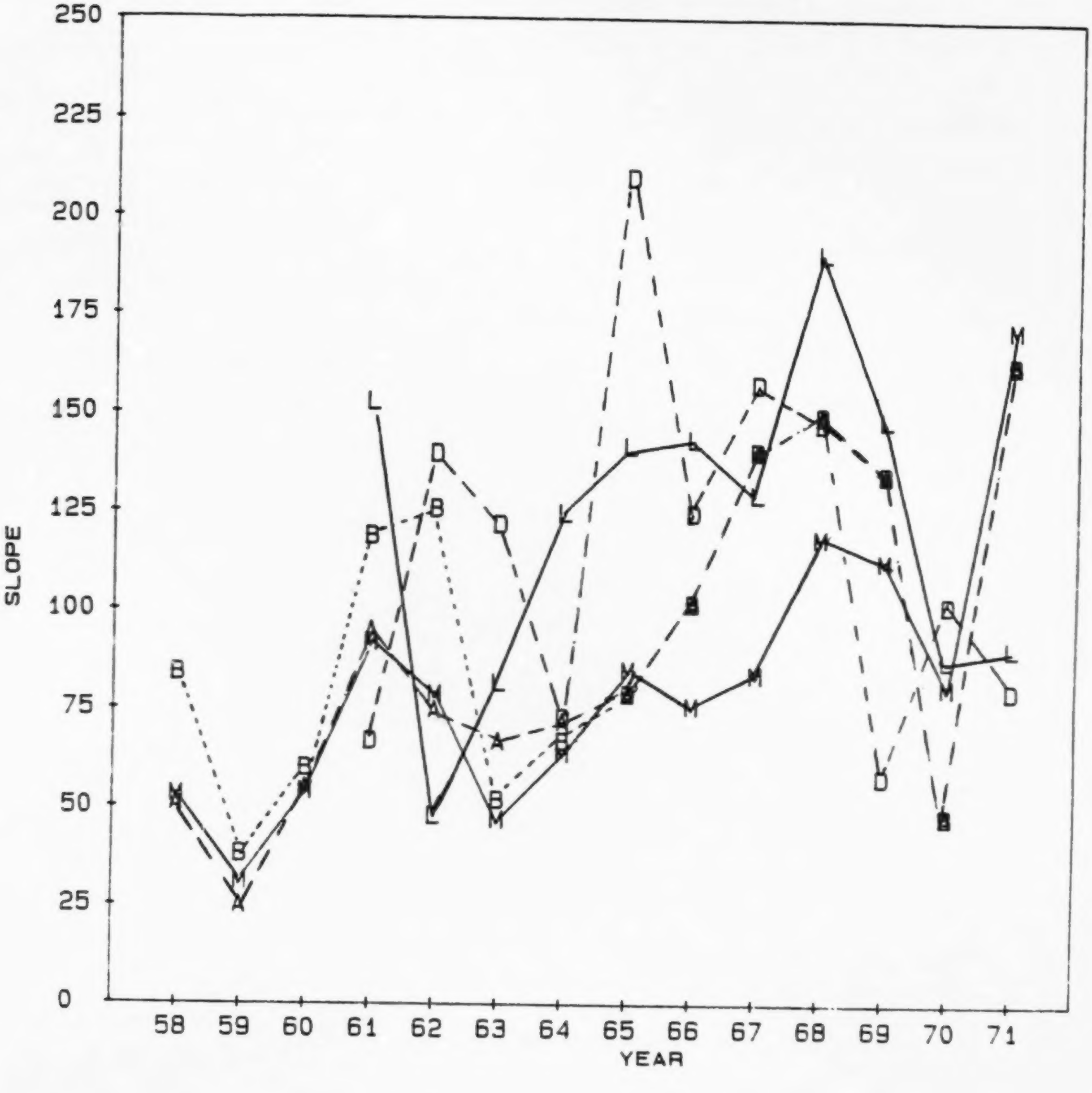

Figure 4. Sensitivity of estimated slope (excess deaths per day per 1000 ug/m^3 BS) to statistical method. Symbols: M, Mazumdar (1982); L, Ostro (1984) for BS < 150; unpublished CARB/EPA analyses: A, all days; B, days with BS < 500; D, days with BS < 200.

Table 1. FEATURES OF REGRESSION MODELS FOR LONDON MORTALITY DATA

Study	Mortality	BS	SO_2	TEMP	Time Series
Mazumdar (1982)*	D15	D15,L	-	D15,L	-
	D15	D15,Q	-	D15,L	-
	D15	-	D15,L	D15,L	-
	D15	-	D15,Q	D15,L	-
	D15	D15,L & Q	D15,L & Q	D15,L	-
Ostro (1984)*	D15	PL150	-	L	-
Shumway (1983)	DW	-	DW, log	DW	Freq., Lag
OTHER ANALYSES					
Ostro (1982)*	D15	PL75	-	L	-
	D15	PL100	-	L	-
	D15	PL200	-	L	-
Shumway (1983)	DW	DW, log	-	DW	Freq., Lag
Ostro (1986)*	C	L	-	L	AR1
	D15	LT200	-	L	-
	D15	LT300	-	L	-
CARB/EPA (1986)*	C	LT200	-	L	AR1-3
	C	-	L	L	AR1-3
	C	LT200	L	L	AR1-3
	D15	LT200	-	L	AR1-3
	D15	-	L	L	AR1-3
	D15	LT200	L	L	AR1-3

*Year by year results.
ARX: Autoregressive model of order x
C: crude
DX: deviations from moving average (X=15 day or weighted)
L: linear
PL_x: piecewise linear with change at x
Q: quadratic
LT_x: linear, truncated above x

Table 2. STATISTICAL SIGNIFICANCE OF SLOPE ESTIMATES RELATING MORTALITY TO BRITISH SMOKE BY YEAR*

	All days		Days with BS < 500	
Year	w/o SO_2	w SO_2	w/o SO_2	w SO_2
1958-59	+	o	+	o
1959-60	+	o	o	o
1960-61	+	+	+	o
1961-62	+	o	+	o
1962-63	+	o	+	o
1963-64	+	+	+	+
1964-65	+	+	+	+
1965-66	+	o	+	o
1966-67	+	o	+	o
1967-68	+	+	+	+
1968-69	+	+	+	+
1969-70	+	o	+	o
1970-71	o	o	o	o
1971-72	+	+	+	o

+: Positive, significant at two-tailed 5% level
o: Not significantly different from zero
-: Negative, significant at two-tailed 5% level.

*Autoregressive model with British Smoke, temperature, relative humidity without SO_2 or with SO_2.

Table 3. RANDOM EFFECTS MODEL FOR DAILY MORTALITY AND BRITISH SMOKE*

	Without Temp. and Humidity in Model		With Temp. and Humidity in Model	
	B	T	B	T
All BS	0.0698	6.71	0.0793	5.83
BS < 500	0.0783	9.55	0.0857	8.74
BS < 200	0.1225	9.57	0.1376	14.64

Table 4. RANDOM EFFECTS MODEL FOR DAILY MORTALITY, BRITISH SMOKE AND SO_2*

	Without Temp. and Humidity				With Temp. and Humidity			
	BS		SO_2		BS		SO_2	
	B	T	B	T	B	T	B	T
All BS	0.0789	3.12	-0.004	-0.296	0.0609	2.72	0.0129	1.06
BS < 500	0.1044	4.95	-0.021	-1.72	0.0839	3.73	0.0025	0.172
BS < 200	0.1352	3.65	-0.012	-0.616	0.1079	3.17	0.112	0.747

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*For all years controlling for year for different levels of British Smoke
A t-statistic $\geq$1.96 is statistically significant at p 0.05.

STATISTICAL REANALYSES OF DATA RELATING MORTALITY TO AIR POLLUTION DURING LONDON WINTERS 1958-1972

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INTRODUCTION

This paper discusses our continuing review and analysis of the London mortality data to assess further the degree of reliance that can be put on the published studies in criteria development and standard setting. We have recently reviewed three statistical analyses of the possible relationship between daily air pollution concentrations and daily deaths in London during the winters of 1958-1971. General conclusions about these studies (Mazumdar et al. 1982; Ostro, 1984; and Shumway et al., 1983) are summarized in the Criteria Document and staff paper addenda and are discussed more fully in a separate memorandum.

We believe that these studies have shown that a relationship exists between mortality in London and particulate matter (measured as British Smoke) and/or SO_2, and that those relationships continue below a British Smoke level of 150 $\mu g/m^3$. However, commentors and others have raised questions about: (1) whether the analyses adequately handled the temporal structure of the data, both in terms of avoiding confounding due to long-term time trends and seasonal fluctuations, and in terms of avoiding the misestimation of the regression standard errors (and hence significance tests) that occurs when there is autocorrelation in the regression residuals; (2) whether the dose-response relationship is linear or nonlinear and whether that relationship is distorted by the techniques used to filter the series, and (3) whether it is British Smoke, SO_2 or both that are responsible for the mortality.

All of the studies have attempted to deal with the autocorrelation in the data (that is, the number of people who die on day t is correlated with the number of people who died on day t-1, t-2, etc.). Some used deviations from 15-day moving averages to remove these autocorrelations; more recently Ostro used an autoregressive model. All of the studies used separate regressions for each year to remove the time trend of falling mortality. None of the models successfully separated SO_2 and British Smoke effects. None of the studies reported any tests to determine whether using separate regressions for each year adequately dealt with the possibility of a linear time trend in the data or reported any formal tests to determine whether they had in fact adequately accounted for the autocorrelations in the data. In addition, while the studies clearly demonstrated relationships at low levels, and gave some indication of their magnitude, no detailed exploratory analysis was presented to determine the potential shape of the dose-response curve.

To address these issues, we joined with analysts at the California Air Resources Board and reanalyzed the London Mortality data to evaluate the adequacy of the fitted models. Following our examination of the underlying data for all years and each year (discussed below), we first studied the temporal structure of the process, particularly its autocorrelation. We computed autocorrelation functions for the dependent and independent variables, ran regressions that accounted for that autocorrelation, diagnosed their residuals to assure that all of the autocorrelation had been controlled for and re-estimated where necessary, examined plots for indications of a relationship and clues to the shape that it might have, and examined the question of whether we could separate the effects of the two variables (i.e. BS vs SO_2).

AUTOCORRELATION FUNCTIONS AND AUTOREGRESSIVE MODELS

First, we examined the autocorrelation functions for mortality, deviations from 15-day moving average mortality, and British Smoke separately for each year 1958-1971. This allowed us to determine whether the data was stationary in each year, to diagnose the nature of the autocorrelation using formal Box-Jenkins techniques, and to determine if the autocorrelations followed the forms used in the examples of Roth et al. (1986) to illustrate the potential for distortion of the dose-response relationship.

Our analyses indicated that in each year, the autocorrelations fell off continuously with increasing lag, showing that the data were stationary. The autocorrelations for both mortality and British Smoke were positive and clearly autoregressive in nature (that is, with a gradual and continuous falling off of the correlation between the value at time t and the value at time t-n as n increased). The autocorrelation function for daily mortality was almost always generated by the first two autoregressive parameters, although for a few years one or three parameters appeared significant. The autocorrelation function for deviations from the moving average of mortality showed greatly reduced levels of autocorrelation. However, even after subtraction of a moving average, enough autocorrelation remained to require an additional autoregressive parameter in order to achieve stationary residuals in most years. The moving average term did induce some short-term cyclical fluctuations in the autocorrelation function that were not previously there; this is consistent with Shumway's finding that a simple 15-day moving average is a filter with short-term oscillations.

In sum, we concluded that the previous analyses were correct in assuming that using separate regressions for each year would achieve stationarity, and correctly assumed that deviations from a moving average, or the use of autoregressive terms would reduce the autocorrelation in the model. However, neither the use of deviations from moving average nor the use of a single autoregressive term seems to completely eliminate the problem, and higher autoregressive parameters are necessary to assure that there is no bias in estimating the significance of the parameters.

At this point it is worthwhile to discuss the problems that autocorrelation in the data can cause. There are cyclical patterns of increase and decrease in both the dependent and independent variables. If those patterns are caused by some omitted factor, including them can induce a false correlation in the data or reduce a true correlation in the data. This requires the omitted factor to be correlated with both mortality and air pollution. The sign of the respective correlations of the omitted factor will determine the direction of the bias. Statistically, if mortality on day t is correlated with mortality on day t-1, etc. then the residuals of the regression of mortality and some independent variables may also be correlated. This violates the classical regression assumption that the errors are uncorrelated, and means that the results of an ordinary least squares regression are not reliable.

It is only autocorrelation in the residuals, and not autocorrelation in the dependent or independent variables, that matters for this problem. The issue of autocorrelation in the series inducing or masking correlations between them is the additional problem of omitted variable bias discussed above. Yet another problem with misspecifying the temporal structure of the model is the misestimation of the regression parameter standard errors, and hence the distortion of significance tests of the parameters.

We performed regressions using an autoregressive model with up to four autoregressive parameters, which remove autocorrelation from the residuals. The residuals of these models were analyzed by standard ARIMA techniques, the regression models were respecified, and repeated, and once again tested to ensure no autocorrelation was present in the residuals.

These regressions were done separately for each year, for dependent variables of daily mortality, and deviation from moving average of mortality. In addition, they were performed in each case with and without controlling for temperature and humidity. Both British Smoke and SO_2 were analyzed as independent variables separately. This gave 112 separate final models (2 dependent variables x 2 pollutants x with or without temperature and humidity x 14 years). In addition, having assured stationarity by performing separate regressions for each year, the overall significance of the results for the full data set was then assessed using a random effects model to incorporate a between year and within year variance in estimating the overall effect. This is described in more detail in the appendix. The results are summarized below.

For daily mortality, British Smoke was significant for 13 out of 14 years, with or without temperature and humidity in the model. The coefficients were generally similar to, but slightly higher than, those reported by Ostro in his regressions using this outcome. In most cases two autoregressive parameters rather than the one used by Ostro were necessary to completely account for the autocorrelation of the residuals. The net effect of removing the remaining autocorrelation that Ostro left in his model was to make British Smoke significant for 2 more years than Ostro found, and to increase the t statistic in all but 3 of the 14 years. This indicates that the temporal patterns in mortality tend to mask rather than enhance the relationship with British Smoke. The coefficients of British Smoke and their t-statistics, for models with and without temperature and humidity, are shown in Table 1. Table 1 also includes the coefficients and t -statistics for the Ostro regressions (which included temperature and humidity).

Note that for both daily mortality and deviations from daily mortality, the regression coefficients tend to increase in the later years, when pollution levels were lower. This is consistent with the results reported by Ostro, who found a higher regression coefficient below 150 than above 150 $\mu g/m^3$ in his spline regressions. The random effects model coefficient for British Smoke was 0.0698 without the weather terms and 0.0793 with them, with t-statistics of 6.71 and 5.83 respectively ($p < 0.0001$). A signed rank test performed to assess the overall significance of British Smoke across all 14 years was highly significant ($p = 0.0011$), both with and without the weather factors.

For deviations from daily mortality, which was more often analyzed in earlier analyses, an autoregressive parameter was necessary in most of the

TABLE 1. DAILY MORTALITY AND BRITISH SMOKE
CONTROLLING FOR AUTOCORRELATION

	Without Temperature and Humidity		With Temperature and Humidity		Ostro Results	
Year	Beta	t-Statistic	Beta	t-Statistic	Beta	t-Statistic
1958	.0487	7.07	.0511	7.06	.062	5.89
1959	.0259	2.88	.0247	2.45	.028	2.34
1960	.0541	3.74	.0549	3.74	.062	2.91
1961	.0849	4.98	.0951	5.05	.093	4.69
1962	.0781	5.81	.0750	5.57	.063	4.08
1963	.0544	3.50	.0669	4.00	.065	3.07
1964	.0726	3.69	.0723	3.49	.065	2.48
1965	.0708	2.45	.0792	2.72	.072	2.40
1966	.1010	3.16	.1017	2.98	.106	3.16
1967	.1046	2.24	.1411	2.94	.227	3.98
1968	.1474	3.27	.1495	3.27	.170	3.34
1969	.1031	2.25	.1346	2.60	.094	1.77
1970	.0505	1.19	.0479	1.12	.066	1.20
1971	.1345	2.26	.1631	2.75	.061	0.79

Mean Coefficient

.0808 .0898 .0881

Signed Rank Test for Overall Significance

S = 52.5 P = 0.0011 S = 52.5 P = 0.0011 S = 52.5 P =.0011

Random Effects Model for All Years

B = .0698 t = 6.71 B = .0789 t = 5.83 B = .0787 t = 5.55

models. However, after its inclusion, British Smoke was significant for all 14 years, with or without the inclusion of temperature and humidity. Six of these years had no days with British Smoke above 500. Coefficients in our random effects model were .0662 (t = 6.49) and .0747 (t = 5.62) without and with temperature and humidity. Note that once autocorrelation is fully accounted for, as in these models, the regression coefficients for using either daily mortality or deviations from daily mortality are quite similar, as one would expect. We conclude that British Smoke is highly significant in this data, after fully accounting for the autocorrelation in the data, whether daily mortality or deviations from daily mortality are used as outcomes. The coefficients of British Smoke and its t-statistic for each year for deviations in mortality are shown in Table 2.

TABLE 2. DEVIATIONS FROM DAILY MORTALITY AND BRITISH SMOKE CONTROLLING FOR AUTOCORRELATION

	Without Temperature and Humidity		With Temperature and Humidity	
Year	Beta	t	Beta	t
1958	.0479	7.39	.0490	6.89
1959	.0412	2.77	.0178	1.61
1960	.0491	3.43	.0529	3.43
1961	.0483	3.47	.0538	3.46
1962	.0626	5.15	.0709	5.68
1963	.0362	2.56	.0527	3.58
1964	.0531	2.85	.0568	2.81
1965	.0914	3.64	.0899	3.33
1966	.0884	3.20	.1055	3.58
1967	.1298	2.62	.1451	2.79
1968	.1180	2.92	.1468	3.79
1969	.1013	2.29	.1130	2.44
1970	.0982	3.02	.0929	2.77
1971	.1294	2.51	.1442	2.66

Random Effects Model for All Years

B = .0662 t = 6.49 B = .0747 t = 5.62

A similar, but not quite as strong pattern is obtained when SO_2 is used as the pollutant. For daily mortality, SO_2 is significant for 10 out of 14 years when temperature and humidity are not in the model, and for 11 out of 14 years when they are included. For deviations from daily mortality, SO_2 is significant for 12 out of the 14 years when temperature and humidity are not included, and for 11 out of 14 when they are included. The coefficients for SO_2 and their t - statistics are shown in Table 3. The random effects model gave weighted coefficients of .0371 without weather terms and .0543 with them ($p < 0.0001$). Again a signed rank test showed a highly significant relationship ($p = 0.0011$) across the years.

DIAGNOSTIC PLOTS AND FUNCTIONAL FORM

The next issue we addressed was the shape of any dose-response relationship, with particular attention to low levels. To examine this, we first plotted the data in various ways. Because pollution accounts for at most a few percent of the mortality in London, and a similar share of its variation, to

TABLE 3. DAILY MORTALITY AND SO_2 CONTROLLING FOR AUTOCORRELATION

Without Temperature and Humidity			With Temperature and Humidity	
Year	Beta	t-Statistic	Beta	t-Statistic
1958	.0644	7.25	.0652	7.35
1959	.0321	2.73	.0325	2.42
1960	.0340	2.30	.0360	2.26
1961	.0714	4.71	.0871	5.24
1962	.0549	7.58	.0541	6.84
1963	.0304	2.40	.0356	2.94
1964	.0388	2.84	.0387	2.65
1965	.0496	2.50	.0820	4.00
1966	.0544	2.85	.0622	2.90
1967	.0296	1.20	.0543	1.93
1968	.0462	1.90	.0588	2.33
1969	.0568	2.57	.0662	2.96
1970	.0281	1.41	.0286	1.37
1971	.0302	1.04	.0542	1.83

Signed Rank Test for Overall Significance

S = 52.5 P = 0.0011 S = 52.5 P = 0.0011

Random Effects Model for all Years

B = .0371 t = 11.24 B = .0543 t = 20.88

detect any curvature in the relationship it is necessary to reduce the variation somewhat by grouping the data in a fashion analogous to that used by Ware et al. (1981) and the 1982 Criteria Document for the 1958-59 London winter data. We examined plots for both outcomes for each year, and for all years combined. We summarized the data in the plots by sorting the observations in order of increasing pollutant, and taking the means of groups.

First, we ran some descriptive statistics on the frequencies of British Smoke at different levels by year. These statistics show that even the early years are dominated by lower levels of British Smoke. In total, only 85 out of the 1540 days in the 14 winters had British Smoke levels above 500 $\mu g/m^3$. In 12 out of the 14 years (all years except 1958 and 1959) over 90 percent of the days were below 500 $\mu g/m^3$. When we did a cut at a lower level of British Smoke we found that 73 percent of the days were below 200 $\mu g/m^3$, including more than 90 percent of the days from 1965 onward, and the majority of the days in 1961 and later.

Figure 1 presents the a scatter plot of daily mortality versus British Smoke, where each point represents the mean of 20 consecutive observations in increasing order of British Smoke. It clearly shows a relationship starting at the lowest observed levels, on the order of 20 $\mu g/m^3$, and also clearly indicates that the slope of the relationship decreases at the highest levels. This is consistent with the results of the published papers. For example, Mazumdar found higher regression coefficients in the later years, when pollution was low, than in the early years, when the average pollution level was much higher. Ostro also reported a higher coefficient below 150 $\mu g/m^3$ than above 150 $\mu g/m^3$ in his spline analysis of the data. The shape of the curve suggests that the log transform used by Shumway should give a better fit than a linear regression. A log transform always has the problem of an infinite slope at the low end. However, since the lowest observed values of smoke were about 20 $\mu g/m^3$, this is unlikely to have been a problem in fitting the regression, although it suggests that the regression should not be extrapolated to values below these.

The same curvilinear shape occurs in the plot of deviations from 15 day moving average mortality against British Smoke (Figure 2). Figure 3 depicts the relationship for British Smoke when only days with smoke levels below 500 $\mu g/m^3$ are included and clearly shows a relationship continuing to the lowest levels. The curvilinear slope is not a function of some change that occurs over time, since it occurs within individual years, as shown in Figure 4, which plots daily mortality versus British Smoke in 1963. The continuation of the relationship to low levels is shown for daily mortality in Figure 5, and for deviation from daily mortality in Figure 6. Note that these plots also provide a second answer to the issue raised by Roth. The plots of the dose-response relationship using deviations from moving average mortality or using mortality as outcome measures both have a similar shape.

The curvilinear shape of the relationship may be an artifact of the autocorrelation of the exposure variable. Since very high pollution days generally follow high pollution days, the population of responders may have been depleted so that it cannot respond proportionately to the very high levels on the following day. In addition the very highest days in 1958 and 1959 (when most of the days over 500 $\mu g/m^3$ occurred) were accompanied by extremely low visibility, and both logic and anecdotal evidence suggests that avertive behavior occurred on those days. However, BS was a larger fraction of particulate mass at higher levels of BS than at lower levels. Such a changing

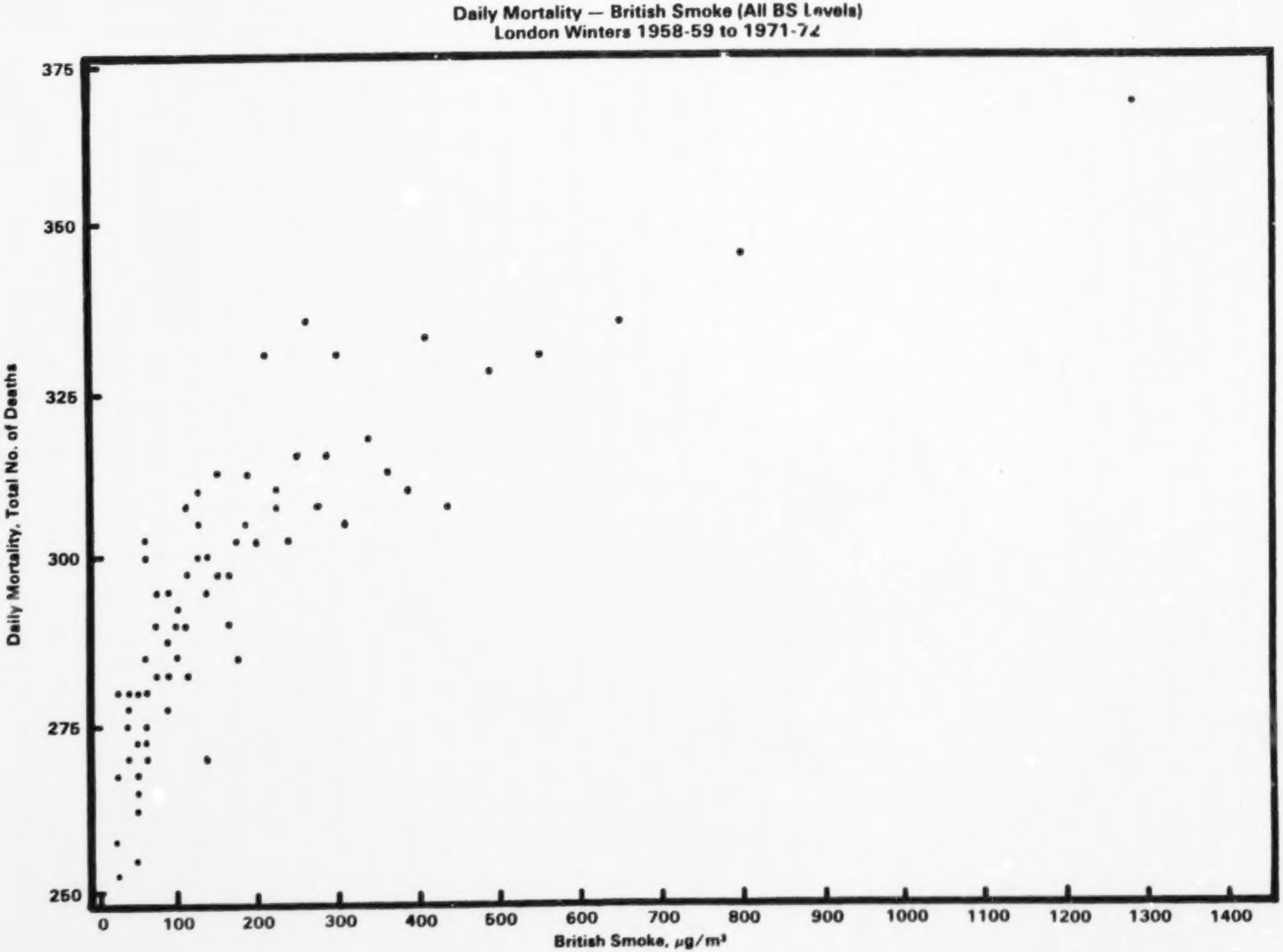
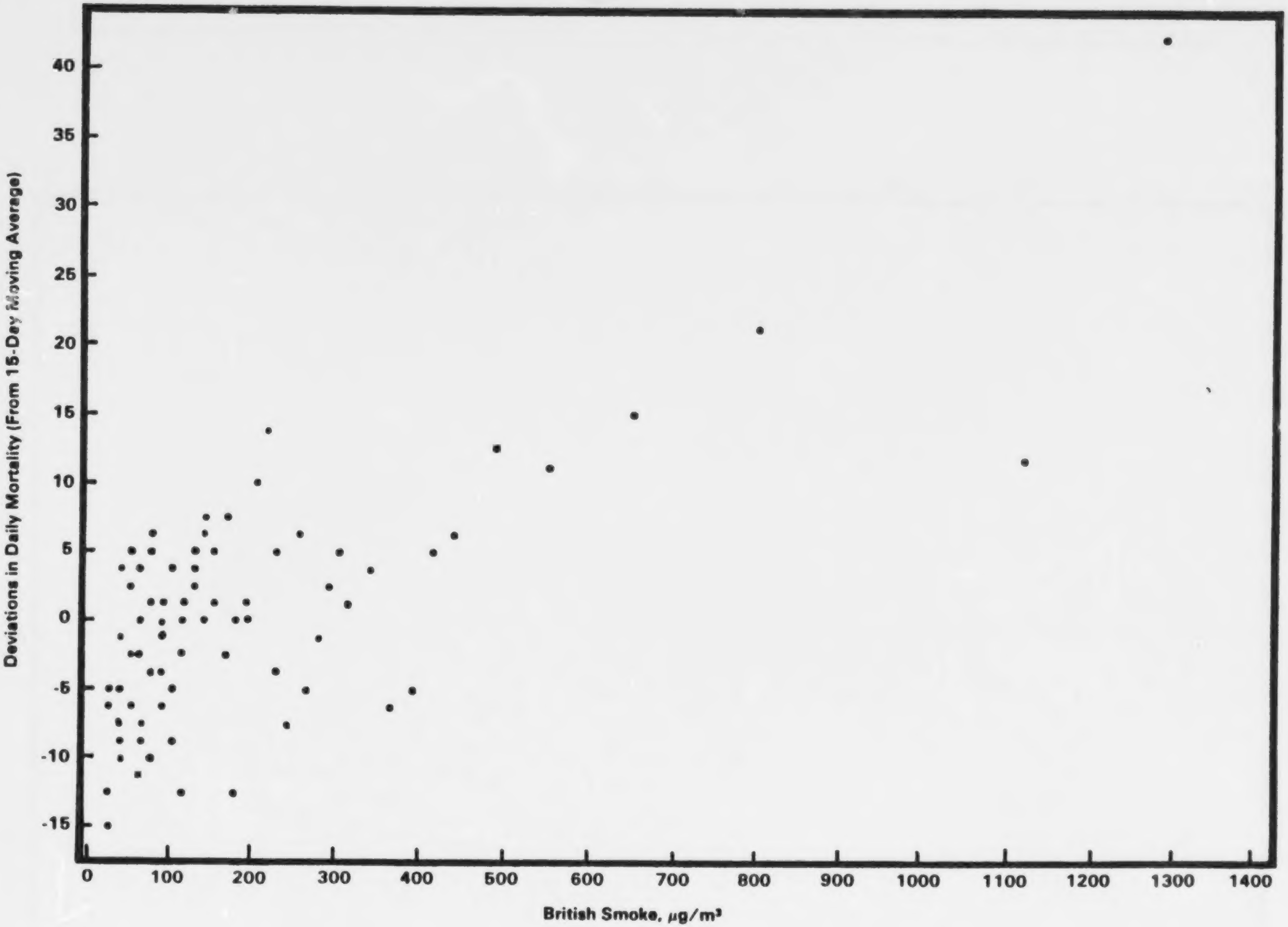


Figure 1. Daily mortality versus British Smoke for the winters of 1958–1972. Each point represents the mean total mortality and mean British Smoke for 20 adjacent values of British Smoke. Note the higher slope at lower BS levels.



Deviations in Daily Mortality vs. British Smoke (All BS Levels)
London Winters 1958-59 to 1971-72

Figure 2. Deviation of daily mortality from a 15-day moving average versus British Smoke for the winters of 1958–1972. Each point represents the deviation mortality and mean British Smoke for 20 adjacent values of British Smoke. Note the higher slope at lower BS levels.

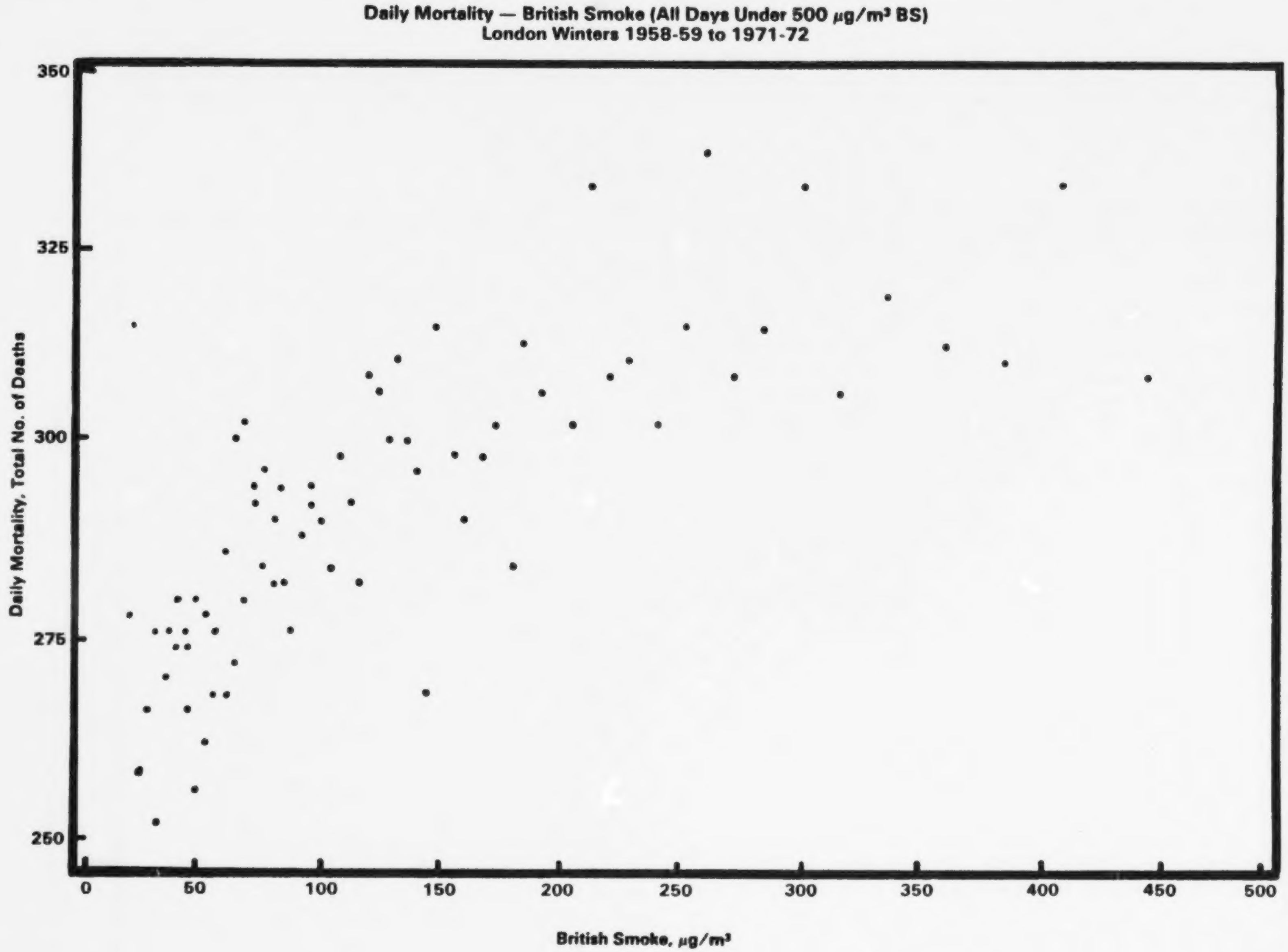


Figure 3. Daily mortality versus British Smoke for the winters of 1958–1972. Each point represents the mean total mortality and mean British Smoke for 20 adjacent values of British Smoke for days with BS < 500 $\mu g/m^3$. Note the relationship continues for levels below 100 $\mu g/m^3$.

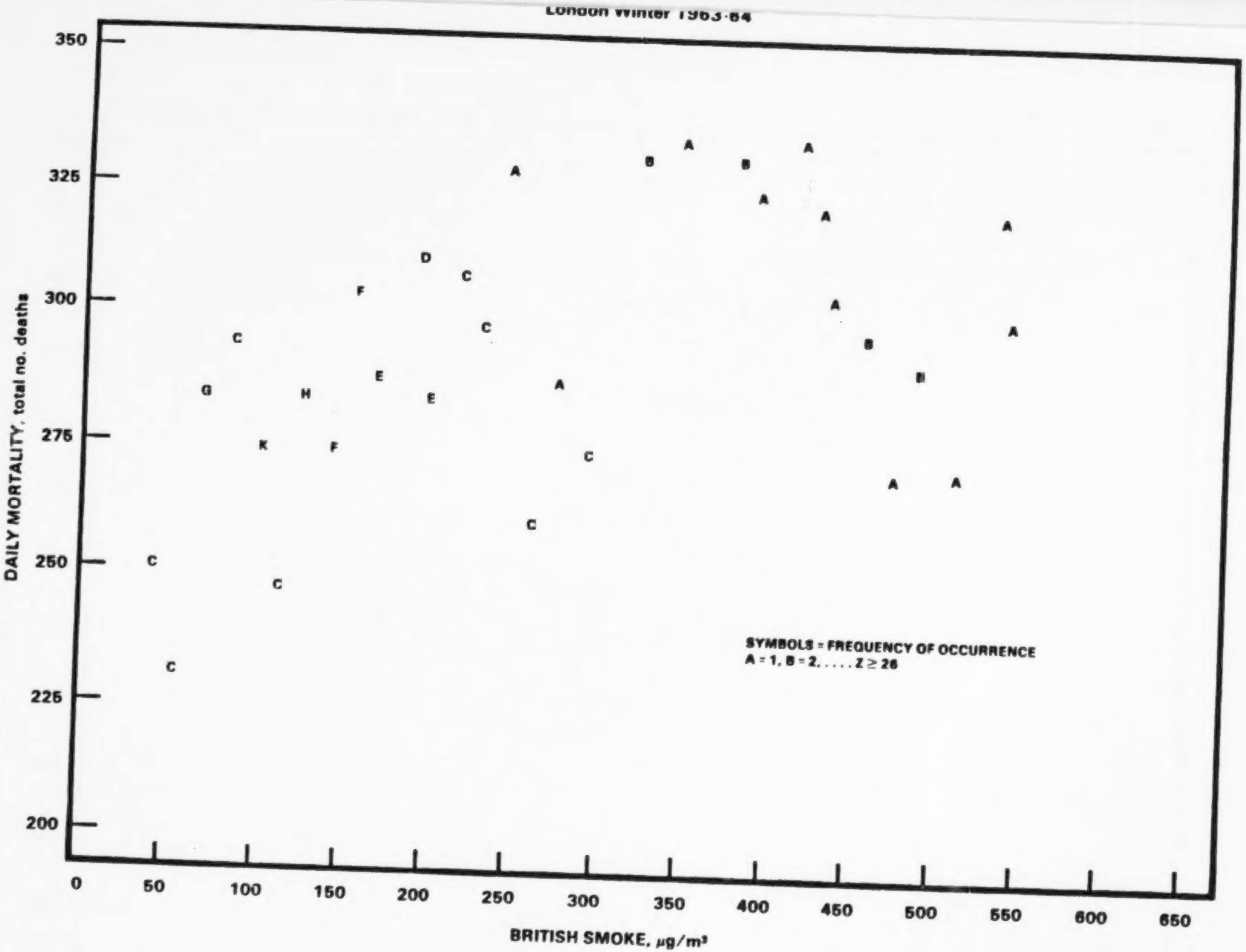


Figure 4. Daily mortality versus British Smoke for the winter of 1963–1964. Each point represents the mean mortality and mean British Smoke level in a 10 $\mu g/m^3$ interval of British smoke. Note the curvilinear response within an individual year.

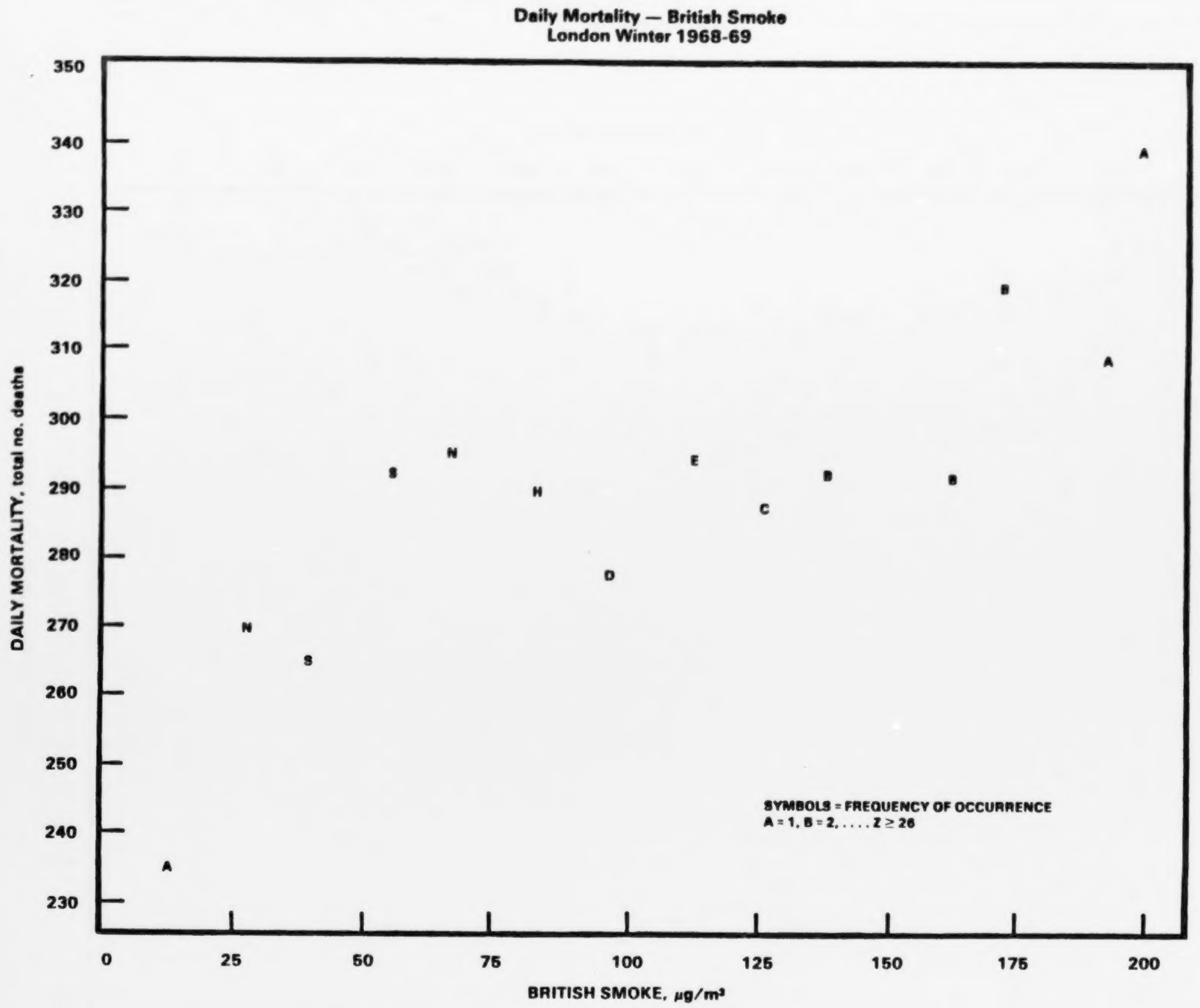


Figure 5. Daily mortality versus British Smoke for the winter of 1968–1969. Each point represents the mean mortality and mean BS level in a

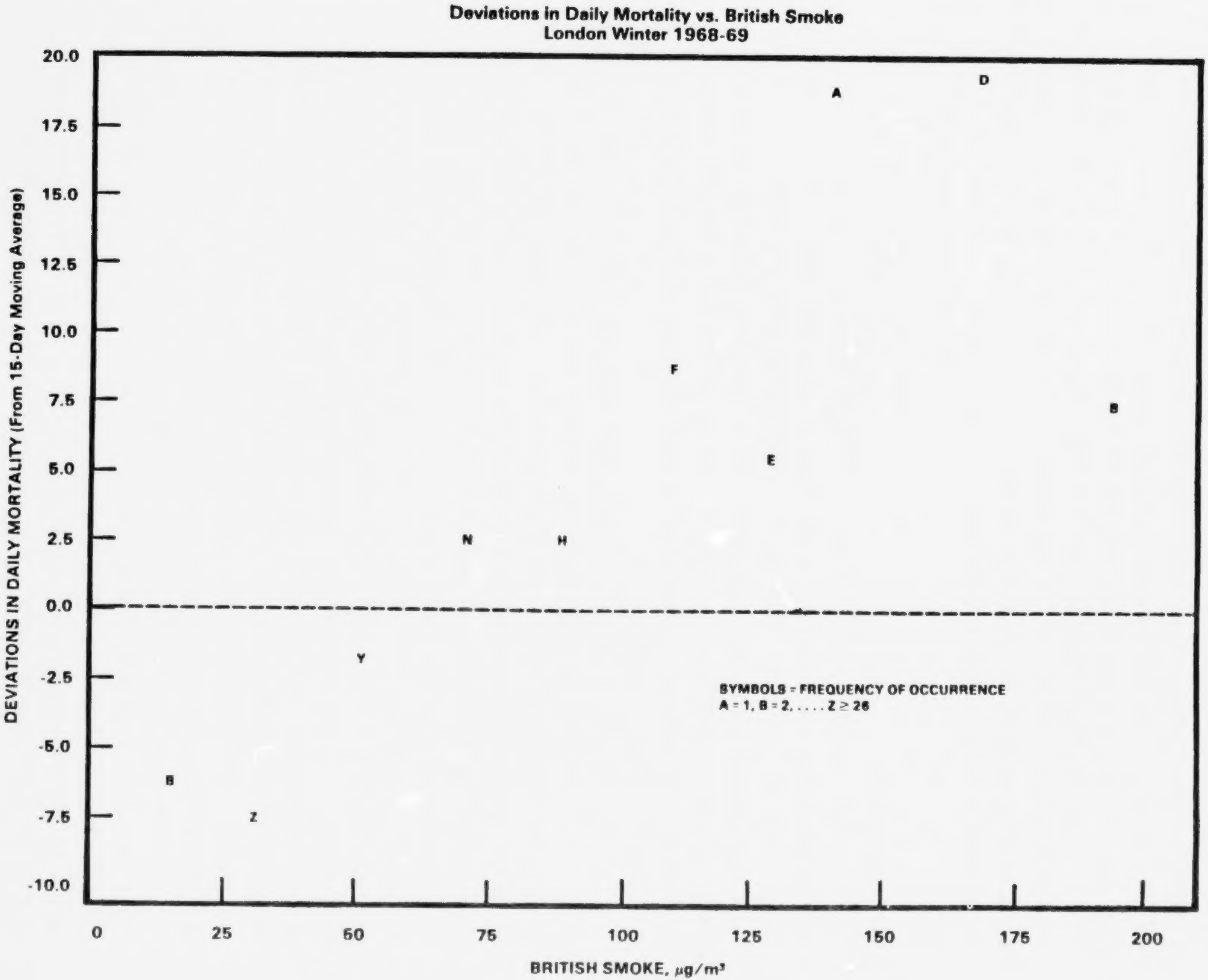


Figure 6. Deviation mortality versus British Smoke for the winter of 1967–1968. Each point represents the mean deviation mortality and mean BS for a 10 μg/m³ interval of British Smoke. Note the linear relationship below 100 μg/m³.

relationship would transform a linear relationship between particulates and mortality into a concave relationship with British Smoke, such as is observed. However, no definitive conclusions as to what caused this curvilinear relationship can yet be made. While examination of the plots suggests that a log or fractional power transformation would fit the data better, that is unnecessary to establish the significance of the correlation. At lower levels, a linear approximation appears to fit almost as well, and we have continued our use of linear models in our subsequent analyses.

LOW LEVEL EFFECTS

Our plots indicated that "hockeystick" regressions would not be appropriate for examining the relationship at lower levels, and that if anything, a higher slope was expected. To examine this quantitatively, we reran all of our regression models using only days when British Smoke was less than 500 $\mu g/m^3$. Since the exclusion reduces the sample size and automatically reduces significance levels, we were more interested in what happened to the slopes than in what happens to the p values.

For daily mortality we found that British Smoke was significant for 12 out of the 14 years, with or without temperature and humidity in the model. SO_2 was significant for 10 out of the 14 years without including temperature and humidity, but for 11 of the years when they were included. For deviations from daily mortality, British Smoke was significant for 11 out of 14 years without including temperature and humidity and for 12 out of 14 years when they were included. SO_2 was significant for 9 out of the 14 years without temperature and humidity corrections, and for 11 out of the 14 years when they were included. However, excluding days over 500 $\mu g/m^3$ reduced the sample size by 40 percent for 1958, giving little statistical power, so that year should probably be excluded from consideration at these levels (which would leave British Smoke significant in 12 out of 13 years for daily mortality and 11 out of 13 for deviations from daily mortality).

Table 4 shows the coefficients of British Smoke and their t - statistics. These show that even with a cutoff of 500 $\mu g/m^3$, the coefficients tend to increase in later years when the pollution levels were lower. This is consistent with Figure 1, which shows a higher slope at lower levels, when only looking at data below 500 $\mu g/m^3$. The exclusion of the days over 500 $\mu g/m^3$

TABLE 4. DAILY MORTALITY AND BRITISH SMOKE (SMOKE < 500 $\mu g/m^3$ ONLY) CONTROLLING FOR AUTOCORRELATION

	Without Temperature and Humidity		With Temperature and Humidity	
Year	Beta	t-Statistic	Beta	t-Statistic
1958	.0774	2.18	.0836	3.47
1959	.0399	1.56	.0375	1.37
1960	.0628	2.70	.0604	2.39
1961	.1080	4.62	.1192	4.96
1962	.1366	4.23	.1264	3.94
1963	.0476	2.50	.0550	2.91
1964	.0680	3.08	.0667	2.82
1965	.0708	2.45	.0792	2.72
1966	.1010	3.16	.1017	2.98
1967	.1046	2.24	.1411	2.94
1968	.1474	3.27	.1495	3.27
1969	.1032	2.24	.1346	2.60
1970	.0505	1.19	.0479	1.12
1971	.1345	2.26	.1631	2.75

Random Effects Model for All Years

B = .0783 t = 9.55 B = .0857 t = 8.74

increases the coefficients in the early years from the values they had when all days were included, which again indicates that the higher slope at lower levels occurs within each year as well as between them and, therefore, is not simply due to some other time trend. Also, note that, even in the years when British Smoke was insignificant, its coefficient was always within the range of the years when it was significant, indicating that the lack of significance was due more to a higher variance in that year, and that a consistent pattern of effect was being seen.

When the random effects model was used, the mean coefficient for British Smoke was .0783 ($t = 9.55$, $p < 0.0001$) without weather terms and .0857 ($t = 8.74$ $p < 0.0001$) with those terms. Note that despite the reduction in sample size, the overall model shows that British Smoke is more significant when restricted to days with pollution less than 500 $\mu g/m^3$ than when the higher days are included. The smaller sample size of the individual winters masks this strength in the individual statistics.

To investigate the low level effects further, the model was rerun using only those days when British Smoke was less than 200 $\mu g/m^3$. While the reduced sample size within each year meant that British Smoke was only significant for

6 out of the 11 years with sufficient data to run the analyses (8 out of the 11 years if one-tailed tests are used), the coefficients were stable, suggesting that using the full data set (which has enough days so that the p-value is not dominated by small sample size) would give a different result. In fact, the coefficient of British Smoke in the random effects model is 0.1225 ($t = 9.57$, $p < 0.0001$) without weather terms and 0.1376 ($t = 14.64$ $p < 0.0001$) with temperature and humidity. Thus the overall relationship between British Smoke and mortality is stronger if the data are restricted to only those days when smoke was less than 200 $\mu g/m^3$. These results are shown in Table 5.

TABLE 5. DAILY MORTALITY AND BRITISH SMOKE (SMOKE < 200 $\mu g/m^3$ ONLY)
CONTROLLING FOR AUTOCORRELATION

	With Temperature and Humidity		Without Temperature and Humidity	
Year	Beta	t-Statistic	Beta	t-Statistic
1961	.1198	1.76	.1207	1.60
1962	.1939	2.88	.1509	2.23
1963	.0436	0.81	.0798	1.51
1964	.1229	2.28	.1242	2.14
1965	.1428	2.68	.1476	2.80
1966	.1436	3.08	.1638	3.28
1967	.1382	2.33	.1861	3.08
1968	.1721	3.75	.1705	3.71
1969	.0786	1.28	.1062	1.67
1970	.0624	1.21	.0455	0.88
1971	.0826	1.18	.1192	1.70

Random Effects Model for All Years

B = .1225 t = 9.57 B = .1376 t = 14.64

To further investigate the relationship between British Smoke and mortality we looked at the years 1965-1972, for smoke levels <200 $\mu g/m^3$. This restricts us to a range of pollution similar to that in the United States today, and a period in London when the sources of the particulate matter were also likely closer to United States sources.

We used a nonlinear regression incorporating all seven years together, with dummy variables for each year, temperature, humidity, and autoregressive parameters. British Smoke was significantly related to mortality ($\beta = 0.138$, $t = 7.67$) with a coefficient that was identical to the one found in our random effects model of the year by year coefficients.

SEPARATING BRITISH SMOKE FROM SO_2

The high degree of collinearity between British Smoke and SO_2 makes it difficult to distinguish between them. Nevertheless we felt that it was important to try, in order to see what could be learned. Reviewing the data above, where only one pollutant was used in the regressions, note that British Smoke was consistently significant more often than SO_2, both in the regressions using temperature and humidity and in those omitting the weather variables, and for both daily mortality and deviations from daily mortality. To examine this further we repeated the above regression for all the ranges of pollution, for inclusion or exclusion of weather variables, and for all years, using both British Smoke and SO_2 in the model. We looked at which pollutant achieved significance and at the stability of the regression coefficients.

For all British Smoke levels, when both pollutants were in the model, British Smoke was a significant predictor of daily mortality for 6 out of the 14 years with or without the weather factors in the model. SO_2 was significant for 3 of the years excluding temperature and humidity, and for 2 of the years if they were included. We then only considered those years when the correlation between the SO_2 coefficient and the Smoke coefficient was less than 0.9. These years have less collinearity and therefore, allow a better chance to distinguish between effects associated with the different variables. This criteria was met for 6 years for the daily mortality regressions that excluded weather. British Smoke was significant for 4 of those years, but SO_2 was not significant for any of them. The criterion was met for 7 years when the weather factors were included; and British Smoke was again significant for 4 of the years, but SO_2 was significant for only 1 year.

When we restricted our models to those days with British Smoke less than 500 $\mu g/m^3$ (Table 6), British Smoke was significant for 7 years and SO_2 for none of the years when temperature and humidity were excluded. When they were included, British Smoke was significant for 5 years and, again, SO_2 was always nonsignificant. Without temperature and humidity, there were 10 years when the correlation of the regression coefficients were less than 0.9, and British Smoke was significant for 5 of them. There were 9 such years when temperature and humidity were included in the model, and British Smoke was significant for 3 of them.

When we looked at the stability of the coefficients, an even stronger story emerged. Using the random effects model, British Smoke was highly

TABLE 6. DAILY MORTALITY, BRITISH SMOKE AND SO_2 ONLY DAYS WITH BRITISH SMOKE < 500 μg/m^3 CONTROLLING FOR AUTOCORRELATION WITHOUT TEMP AND HUMIDITY

Year	BS	t	SO_2	t
1958	.0646	1.28	.0260	.367
1959	.0876	2.08	-.0638	-1.43
1960	.0652	1.44	-.0028	-.061
1961	.1691	2.83	-.0482	-1.07
1962	.0950	1.50	.0345	.761
1963	.1173	2.93	-.0622	-1.95
1964	.1357	2.34	-.0467	-1.26
1965	.0299	.478	.0315	.741
1966	.1071	1.36	-.0040	-.086
1967	.2085	2.36	-.0628	-1.37
1968	.2147	2.84	-.0436	-1.11
1969	-.0462	-.375	.0773	1.30
1970	-.0030	-.036	.0294	.745
1971	.2699	2.40	-.0769	-1.40

Random Effects Model for all Years

BS = .1044 t = 4.95 SO_2 = -.0212 t = -1.76

significant (β = 0.0789, T = 3.12). Using daily mortality as the outcome, for British Smoke at all levels, the mean BS coefficient with SO_2 in the model was 0.089, compared to a mean of 0.081 when only British Smoke was in the model. The impact of adding SO_2 to the model was only to change the variation about that mean, with the coefficient of variation increasing from 43 percent to 105 percent. A sign rank test of the overall significance of British Smoke over the full 14 year period was still significant (p = 0.0011). By contrast, the mean value of the SO_2 coefficient became negative in the joint models, and its coefficient of variation increased to 750 percent. The sign rank test for an overall effect was highly insignificant (p = 0.66).

When temperature and humidity were included in the models, the mean value of the British Smoke coefficient was 0.0737, with a coefficient of variation of 119 percent and signed rank test p-value of 0.0144. This compares to a mean of 0.090 and COV of 47 percent without SO_2 in the model. For SO_2, the mean coefficient was 0.0111 with a coefficient of variation of 440 percent and a signed rank test p-value of 0.47. This compared to a mean of 0.054, COV of 34 percent, and overall significance level of p = 0.0011 without British Smoke in the model. In the random effects model, British Smoke was again significant (β = 0.0609, t = 2.72) and SO_2 nonsignificant.

When both pollutants were included in models for only those days when British Smoke was less than 200 μg/m^3 (Table 7), and analyzed in the random effects model, the coefficient of British Smoke was 0.1352 ($t = 3.65$ $p = .0019$) without temperature and humidity terms, and 0.1079 ($t = 3.17$ $p = 0.0045$) with those terms. SO_2 was highly nonsignificant in both cases. This compares to the equivalent coefficients in models excluding SO_2 of 0.1225 and 0.1376 respectively, as noted previously. Again, the coefficient of British Smoke is little changed by the addition of SO_2 to the model.

TABLE 7. DAILY MORTALITY, BRITISH SMOKE AND SO_2 FOR ONLY DAYS WITH SMOKE < 200 μg/m^3 CONTROLLING FOR AUTOCORRELATION WITHOUT TEMP AND HUMIDITY

Year	BS	t	SO_2	t
1961	.0485	.417	.0588	.754
1962	.1006	.883	.0583	1.03
1963	.1215	1.52	-.0723	-1.31
1964	.1274	1.30	-.0034	-.056
1965	.1850	1.97	-.0277	-.538
1966	.1683	1.64	-.0135	-.271
1967	.2517	2.54	-.0687	-1.40
1968	.2472	3.27	-.0483	-1.26
1969	-.1879	-1.19	.1278	1.81
1970	.0225	.247	.0214	.534
1971	.2070	1.65	-.0648	-1.18

Random Effects Model for All Years

$B = .1352$ $t = 3.65$ $B = -.0117$ $t = -.616$

The stability of the British Smoke coefficient to the addition of SO_2 to the model, the fact that it remains significant in about half of the individual years and in the analysis of all years in contrast to the instability of SO_2, and the general nonsignificance of SO_2 both in individual years and overall, suggests different conclusions about the two variables. The multiple regression with both factors included looks for significance just for that portion of British Smoke and SO_2 that vary independently of each other. For British Smoke, we find that the coefficient, so restricted, is the same as when all of its variance is considered (in the regressions with only one pollutant). Moreover, this effect was statistically significant over the whole data set. SO_2, in contrast, had a different mean coefficient when only its variation that was independent of smoke was considered, and that coefficient was not significantly different than zero. This suggests that the significance of SO_2 in separate regressions may only be due to its collinearity with British Smoke, whereas British Smoke is significant, with the same magnitude effect, whether or not its covariance with SO_2 is included. Because of the high degree of collinearity, this cannot be used to exclude the possibility of an independent SO_2 effect; however, we feel that it is good grounds for concluding that British Smoke is significantly correlated with mortality independent of SO_2.

The year by year analysis serves an additional purpose besides controlling for possible linear trends in the data. It also serves as a partial control for omitted variable bias. Any epidemiological study always faces the issue of the omitted confounding variable. Given that British smoke is significant in 13 out of 14 years taken individually means any omitted factor accidentally linking particulate matter and mortality cannot be a happenstance but must be a long-term systematic factor. It is of course possible to imagine such factors, such as weather, which was controlled for, but perhaps imperfectly. However what makes this relationship so impressive is that it was so stable across a period of 14 years when the nature of air pollution in London was changing drastically.

In the 1950's and early 1960's particulate matter was dominated by the open hearth burning of coal, which was banned by the Clean Air Act of 1963. The growth in diesel bus and truck traffic in London combined with the fall in open combustion substantially changed the source and weather-sensitive nature of the particulates. For example in the first 4 years (1958-1961) the average correlation coefficient of British Smoke with temperature was -.300 and with relative humidity was +.325. In the last 4 years of our data they had fallen to -.188 and +.084 respectively. Given that when we restrict our regressions to the linear end of the dose-response curve (BS $<200\ \mu g/m^3$) the coefficients are stable from the beginning years to the end ones, while the sources of particulate matter and their relationship to weather change significantly, such omitted variable bias seems unlikely.

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APPENDIX

RANDOM EFFECTS MODEL FOR ESTIMATING AN OVERALL RELATIONSHIP BETWEEN POLLUTION AND MORTALITY FOR ALL YEARS

We assume a model where;

b_i is estimated separately for each year, with variance V_i

and residual e_i

the random effects model is that

$b_i = B + e_i + er$ where er is a random variance component with variance Vr

then we can estimate Vr by

$$Vr = [\Sigma\ (b_i - bavg)^2]/(k - 1) - \Sigma\ V_i/k$$

and

$$B = \Sigma\ w_i b_i / \Sigma\ w_i$$

where

$$w_i = 1/(i + VR)$$

and

$$\text{se of } B = [\Sigma\ W_i]-0.5]$$

☆U.S. GOVERNMENT PRINTING OFFICE : 1987-748-121/40694